

June 1993

**EVALUATION OF DELISTING CRITERIA AND REBUILDING  
SCHEDULES FOR SNAKE RIVER SPRING/SUMMER  
CHINOOK,  
FALL CHINOOK, AND SOCKEYE SALMON**

Recovery Issues for Threatened and Endangered Snake River Salmon  
Technical Report 10 of 11

Technical Report 1993



This report was funded by the Bonneville Power Administration (BPA), U.S. Department of Energy, as part of BPA's program to protect, mitigate, and enhance fish and wildlife affected by the development and operation of hydroelectric facilities on the Columbia River and its tributaries. The views of this report are the author's and do not necessarily represent the views of BPA.

This document should be cited as follows:

<i>Cramer, Steven P.; Doug Neeley, U.S. Department of Energy, Bonneville Power Administration, Division of Fish and Wildlife, Project No. 1993-013, Contract No. DE-AM79-1993BP99654, 231 electronic pages (BPA Report DOE/BP-99654-10)</i>
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**EVALUATION OF DELISTING CRITERIA AND  
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**Recovery Issues for Threatened and Endangered Snake River Salmon  
Technical Report 10 of 11**

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**U.S. Department of Energy  
Bonneville Power Administration  
Division of Fish and Wildlife  
P.O. Box 3621  
Portland, OR 97208-3621**

**Project No. 93-013  
Contract No. DE-AM79-93BP99654**

**June 1993**

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## ACKNOWLEDGEMENTS

Deborah Watkins served as Project Manager for Bonneville Power Administration on this report. Deborah was especially helpful in locating and distributing large quantities of resource information to our consultant team. Steven Vigg of Bonneville Power Administration had the vision to assemble a consultant team capable of tackling a diversity of topics that should be integrated in recovery planning for Snake River salmon. We are grateful for Steve's initiative in setting up this project and for his counsel in guiding its progress.

Erick Van Dyke of our staff reviewed the entire ESA administrative record on the Snake River salmon stocks, and was instrumental in screening that voluminous set of reports and letters to obtain the most current information for our purposes. Erick also prepared many of the graphs and tables presented in the report.



## EXECUTIVE SUMMARY

In this report, we develop a framework for distinguishing healthy and threatened populations, and we analyze specific criteria by which these terms can be measured for threatened populations of salmon in the Snake River. We review reports and analyze existing data on listed populations of salmon in the Snake River to establish a framework for two stages of the recovery process: (1) defining de-listing criteria, and (2) estimating the percentage increase in survival that will be necessary for recovery of the population within specified time frames, given the de-listing criteria that must be achieved. We develop and apply a simplified population model to estimate the percentage improvement in survival that will be necessary to achieve different rates of recovery.

We considered five main concepts identifying de-listing criteria: (1) minimum population size, (2) rates of population change, (3) number of population subunits, (4) survival rates, and (5) driving variables. In considering minimum population size, we conclude that high variation in survival rates poses a substantially greater probability of causing extinction than does loss of genetic variation. Distinct population subunits exist and affect both the genetic variability of the population and the dynamics of population decline and growth. We distinguish between two types of population subunits, 1) genetic and 2) geographic, and we give examples of their effects on population recovery.

In order to be de-listed, we recommend that the spring/summer and fall chinook populations should meet at least one of two criteria:

- 1) abundance of populations in good or excellent habitat should show a significant exponential increase for at least two generations. Simulations indicate that a minimum of two generations are required for the population to begin to stabilize. Populations in fair habitat should be stable for at least two generations.
- 2) spawner abundance or parr density should average 70% of carrying capacity for one generation. Characteristics of the stock-recruitment relationship were used to derive this criterion.

We recommend that delisting criteria for the sockeye population be developed from

information gathered as rebuilding progresses.

## EXTINCTION MODELS

We reviewed the extinction model that NMFS used as a basis for listing. The model is very sensitive to extinction number, and extinction was defined by NMFS in the model as the return of only one fish. A slight increase in the extinction number dramatically increased the estimated probability of extinction. In practical terms, the extinction number should correspond to the point at which all fish would be taken as captive broodstock to prevent extirpation. We provide guidance as to how that number should be determined.

NMFS included a measure of year-to-year trend variation in the model, but it did not include variability in the parameter estimates in estimating the extinction probabilities. This source of variability increases the estimated extinction probabilities considerably. We also evaluated NMFS's use of running totals over years and its choice of lag time as model input.

## TREND MEASURES

During the initial part of a successful recovery program, exponential models would be appropriate tools to characterize the rebuilding trend. We evaluated three different trend estimators of simple exponential growth: Least squares regression of the log of spawner abundance on time; the log of the abundance ratio of the return spawner (recruit) to brood-year spawner (parent); and the Poisson regression of untransformed abundance on time. The Poisson regression seems to give the most accurate estimate of abundance over a period of exponential decline or growth. Although this tool will not be familiar to most biologists, it is one worth considering for estimating trend. The log of the spawner recruit-to-parent ratio is the least precise of these three estimators of trend. However, the recruit-to-parent ratio is the most biologically meaningful of the three measures. It should be used in conjunction with a more powerful tool such as Poisson regression.

The Ricker function can be used to estimate the point of maximum growth or maximum sustainable production. However, the estimate is not very precise and statistical tests regarding its hypothesized value are not very powerful if derived from data during periods of exponential growth. This function would probably not be the appropriate tool to

assess trend during the early part of the recovery program; however, it would be a valuable tool to apply when the growth in the population slows and its estimate becomes more precise.

## SPRING/SUMMER CHINOOK

We compared existing estimates of the stock-recruitment parameters of a Ricker function for spring/summer chinook and adjusted them to pristine survival rates. Throughout this report, we use the terms "pristine" and "pre-dam" in reference to the survival juvenile and adult salmon (excluding harvest) would have experienced during their migration through the mainstem Snake and Columbia rivers prior to the development of mainstem dams and hatcheries. The net result of these adjustments to pristine migration conditions was that values of  $\alpha$  ranged from 10.5 for the Columbia River to 16.9 for the Deschutes River. We concluded from this comparison that an  $\alpha$  value of 12 for pristine conditions was a reasonable approximation for spring/summer chinook in the Snake River. We estimated the  $\beta$  value of the Ricker curve for Snake River spring/summer chinook from the carrying capacity of the basin for smolts. The stock-recruitment function for the Snake River spring/summer chinook indicates that the pristine population (with today's carrying capacity) would have had a maximum recruitment of 500,000 chinook produced by about 200,000 spawners. Maximum sustainable yield would have been about 330,000 fish produced by about 150,000 spawners. Harvest and mortality rates more than 80% above the levels under pristine conditions will cause the surplus production to drop steeply. Population collapse is reached when the harvest and mortality rates added to pristine levels reach slightly above 85%.

To demonstrate the possible influence of geographic subunits, we divided the population into four subunits corresponding to the four habitat quality ratings used by IDFG (1992). We chose Ricker  $\alpha$  and  $\beta$  parameters for each of the four subunits such that when the subunits were combined, they produced an overall population with similar parameters to those estimated for the population as a whole. In the absence of down-stream and up-stream passage mortality, the harvest rate at which MSY is achieved is 56.5%, 67.5%, 78.7%, and 84.4% for the poor, fair, good, and excellent habitat subunits, respectively. These differences indicate that MSY for the excellent habitat subunit could not be achieved without substantially over-harvesting the poor and fair habitat subunits.

We adjusted Raymond's (1988) estimates of smolt-to-adult survival for wild yearling

chinook to account for smolt passage mortality, and found that similar adjustments to other data sets resulted in a high level of agreement on smolt-to-adult survival rates with the analyses of Petrosky (1991), Lindsay et al. (1989), and Fast et al. (1991). We estimated the smolt-to-adult survival under pristine conditions averaged 10.4%.

We found sufficient evidence of bias in the reconstructed count of wild fish, and we do not believe that the historical record at Lower Granite Dam will serve as pre-recovery base for comparing future counts. If the clipped adipose fin is desequestered, then the dam count could be an excellent measure of basin-wide natural/wild escapement.

Historic redd counts cannot be used to estimate absolute abundance because the index areas are not representative of the whole of the spawning habitat. We recommend stratified random sampling be used to choose redd count areas for the purpose of estimating future abundance at the subpopulation and population level. However, redd counts should be continued in the index areas to estimate trend. The index redd counts provide the best measure of trend available. There are indications that the mean trend over index areas may be a more powerful measure of trend than the pooled total count.

Parr density is probably the best measure of population health. The parr densities are well below what would be expected of a healthy population. The number of years over which consistent parr density surveys have been conducted is very limited, therefore a precise assessment of parr-density trend is not yet possible. In the future, redd and parr sampling frames should be integrated.

The smolt passage index may provide a good indicator of smolt abundance if it can be adequately adjusted for years of high levels of spill. There is evidence that the index is not consistent between years of little spill and years of high spill. A method of adjustment based on the proportion of the passage that is of hatchery origin is presented. If the method proves to be appropriate, then an index of natural adult-return to natural smolt survival based on Lower Granite passage may be possible.

There are indications of a decrease in the summer proportion of the Ice Harbor Dam spring/summer dam counts. To what extent this has affected the relative summer distribution in wild fish is not known, but we found no indication of a change.

Even though age distribution is highly variable over time, there are differences in the age distributions both within and among subbasins. Age distribution is an important

measure for allocating brood-years to parents for the purpose of developing spawner based recruit-to-parent ratios.

Future spawner surveys should be based on counts of spawned-out carcasses, not just based on the number of dead fish. Age distributions should be summarized for each sex, and the age distribution should be based on scale analysis not length of fish. This standard should be followed by all agencies involved in evaluating the recovery program.

## FALL CHINOOK

We compared existing estimates of the stock-recruitment parameters of a Ricker function for fall chinook and adjusted them to pristine survival rates. The adjusted  $a$  from Schaller and Cooney (1992) for Snake River fall chinook (19.2) was close to the upper range for  $\alpha$  estimated by other workers, so we used  $\alpha = 20$ . We assumed that production of fall chinook was limited by spawning habitat, such that, the maximum number of recruits that could be produced was determined by the maximum number of spawning pairs the basin could support. We analyzed two values of  $\beta$ , one corresponding to habitat remaining in the Snake River only, and the other including habitat in the Clearwater Basin. Schaller and Cooney (1992) estimated that in the mainstem Snake River from Hells Canyon Dam down to Lower Granite Dam, 4,800 fall chinook spawners would produce maximum recruitment. Arnsberg and Connor (1992) estimated that the Clearwater River below the confluence of the North Fork alone could support 191,000 fall chinook spawners, and it is likely that substantial additional habitat suited to fall chinook exists elsewhere in the Clearwater Basin.

The stock-recruitment function for fall chinook indicated that once the harvest rate and mortality rates exceeded the pristine level by about 80%, the sustainable yield (or surplus production) dropped steeply. This indicates that only a few percentage points change in mortality separates a population in collapse from a population near carrying capacity.

We estimated passage mortality of juvenile fall chinook averaged 22% per dam, and we used this to back-calculate pristine smolt-to-adult survival rates based on estimates for fall chinook in the mid-Columbia. This method gave an estimate of 12.6% for smolt-to-adult survival rate before dams were built. The high smolt-to-adult survival rate for fall chinook was reflected, in part, in their capacity to produce more recruits per spawner than spring

chinook.

## **SOCKEYE**

Because the sockeye population is functionally extinct in the wild, it is anticipated there will be at least two generations, or after 1998, before significant numbers of adults will spawn naturally. Therefore, we did not estimate stock-recruitment parameters, but rather, we recommended that the monitoring program for smolt abundance and adult recruits be used as the basis for estimating the stock-recruitment relationship of Redfish Lake sockeye as the stock rebuilds.

## **SIMULATION OF REBUILDING**

### **Spring/Summer Chinook**

We used simulations to identify ranges of recovery times that might be expected to achieve the recommended de-listing criteria. We repeated the simulations for a single homogeneous population and for four population subunits corresponding to habitat quality. The simulations with population subunits provided a better fit to the observed trends, but still over-reacted to the assumed increase in mortality rates at dams. These simulations indicated that even under baseline conditions, the population would increase slightly, and that each increment of improved survival would allow a larger population to be maintained. The simulation with population subunits demonstrated that a 20% increase in survival would cause about a 60% increase in the overall population above that for baseline conditions, but the population subunit in poor quality habitat would still go extinct and the subunit in fair habitat would still decline.

In order to achieve the 70% of capacity criterion for recovery, an escapement of about 50,000 fish into good and excellent habitat would be required. A 20% to 40% increase in survival rate would be necessary to maintain a stable population in fair habitat and reach 50,000 spawners in good and excellent habitat. Further, the simulations suggest that the escapement criterion for de-listing could be achieved in about 15 years with a 40% increase in survival and about 10 years with a 75% increase in survival.

**Fall Chinook**

The simulated returns of fall chinook closely matched the observed declining trend in wild spawners. The simulations indicated that under existing conditions, the Snake River population would remain constant at about 400 fish, but if habitat were seeded in the Clearwater Basin, the population would gradually increase to over 2,500 fish within 50 years. Fall chinook were eliminated from the Clearwater Basin in 1927 when their passage was blocked by Lewiston Dam, but Lewiston Dam was removed in 1973 and the basin is now fully accessible to fall chinook. The run size to the Snake River was predicted to be about 40 times greater if habitat in the Clearwater Basin was used. The simulation in which survival rate of fall chinook was increased 40% resulted in the population stabilizing after about 15 years at just above the spawner level required for 70% of MSY. Thus, a 30% to 40% increase in survival is the minimum that would achieve our de-listing criteria based on spawner abundance. Our simulations indicate that either a 60% or 75% increase in survival would enable the minimum spawner criterion to be achieved in two generations.

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EVALUATION OF DE-LISTING **CRITERIA** AND REBUILDING SCHEDULES  
FOR SNAKE **RIVER**  
SPRING/SUMMER CHINOOK, FALL CHINOOK, AND SOCKEYE SALMON

**1. INTRODUCTION**

1.1 RECOVERY **PLANS** AND DE-LISTING **CRITERIA**

Snake River sockeye salmon, fall chinook salmon, and spring/summer chinook salmon populations are severely depleted and have been listed as threatened or endangered under the Endangered Species Act (ESA). A stipulation of the ESA is that NMFS must develop a Recovery Plan for the conservation and survival of endangered and threatened species (ESA, Section 4(f)). To the maximum extent practicable, a recovery plan must contain (1) a description of site-specific management actions to achieve the plan's goal for the conservation and survival of the species; (2) objective and measurable criteria which, when met, result in the removal of the species as listed; and (3) estimates of the time required and cost to carry out these measures. Identification of the biological elements of items (2) and (3) is the focus of this report.

"Healthy" and "threatened" populations must be defined in such a way that the difference between the two can be readily measured. Thompson (1991) thoroughly reviewed the literature on conservation biology and found that there was not a strong consensus on how to measure population viability. In this report, we develop a framework for distinguishing between healthy and threatened populations, and we analyze specific criteria by which these terms can be measured for threatened populations of salmon in the Snake River.

**1.2 CRITERIA USED TO QUALIFY SNAKE RIVER SALMON FOR LISTING**

One of the approaches used by NMFS to determine that Snake River spring-summer and fall chinook stocks should be listed was to estimate probabilities of extinction using an exponential diffusion model detailed in Dennis *et al* (1991). The model utilizes the mean and variance of the instantaneous rate of change in population size over time. NMFS applied historical data to this model to estimate the probability of extinction within a specified period. Further, NMFS has used these probability distributions to estimate population size threshold values (which they refer to as Minimum Viable Population Size (MVP)). These threshold values are used to separate listing classifications based on Thompson's (1991) definition:

Endangered:           At least a 5% percent chance of extinction over the next 100 years.

Threatened:           At least a 50% change of becoming endangered within 10 years.

Thus, the question posed by NMFS to determine if a stock should be listed is, "If nothing is done, are the stocks likely to be here in another 100 years?" The choice of the probability and the period of time is somewhat arbitrary but was intended to reflect a time frame that can be reasonably addressed by a recovery effort.

Probability models were not used in assessing whether Snake River sockeye should be listed as "endangered" or "threatened", nor were such models necessary. An escapement of four returning adults (three males and one female) in 1991 are definitely endangered and may effectively be extinct. After a review of the Thompson (1991) paper and conservation biology literature, Allendorf (1991) concluded that populations of 1,000 or less should generally be considered "endangered", and that it takes population sizes on the order of 10,000 to be considered healthy.

Some recovery programs (e.g., U.S. Fish and Wildlife's Colorado Squawfish recovery program, 1978) have two stages in their reclassification program for species/stocks that are **classified** as endangered:

Downlisting step: A reclassification of the species from "endangered" to

"threatened" status based on one set of criteria.

De-listing step: The reclassification of the species from "threatened" to "non-threatened".

We see little benefit to be gained by adding a downlisting step. In this document we will only discuss de-listing as it applies to the complete removal any listed stock, be it classified as "endangered" or "threatened".

### 1.3 APPROACH AND SCOPE OF THIS REPORT

In this report we review reports and analyze existing data on listed populations of salmon in the Snake River to establish a framework for two stages of the recovery process: (1) defining de-listing criteria, and (2) estimating the percentage increase in survival that will be necessary for recovery of the population within specified time frames, given the de-listing criteria that must be achieved. We begin first by identifying the characteristics of an ideal criterion for de-listing. We then synthesize the pertinent data and analyses on parameters of Snake River salmon stocks, evaluate those parameters in terms of the characteristics desired for de-listing criteria, and recommend the de-listing criteria which will be most efficient for each of the listed populations of Snake River salmon. We use the term "efficient" in reference to the effectiveness of a criterion at detecting population health at the earliest time possible during the recovery process. Population parameters which we consider include habitat carrying capacity, stock-recruitment parameters, abundance measures (spawners, juveniles, and adult recruits), and survival measures between life stages (parr-to-smolt, dam passage, smolt-to-adult, harvest rates, inter-dam loss of adults). After reviewing these parameters and recommending de-listing criteria, we develop a simplified population model for the life cycle of each of the listed populations and we use simulations to estimate the percentage improvement in survival that will be necessary to achieve different rates of recovery.

## 2. DE-LISTING CONCEPTS

There are at least four main concepts that we believe should be considered in any set of de-listing criteria: (1) minimum population size, (2) rates of population change, (3) number of population subunits, and (4) survival rates.

### 2.1 POPULATION SIZE AND POPULATION CHANGE

The minimum viable population (MVP) size has been recommended as a basis of de-listing in some recovery programs. The MVP criterion for de-listing would be a minimum population size giving an acceptable probability of persistence over a given a period of time. For example, the U.S. Fish and Wildlife Service (USFW) has established the following de-listing criteria for the Cui-ui (1991):

*De-list when the chance of persistence over the next 200 years is 95% (an extinction probability of 5% over 200 years).*

Estimation of MVP values are usually developed for species that are in decline. MVP values based on estimated parameters using historic population data over the period of a decline would not be applicable to a recovery period when the population is increasing. If the population size required to guarantee a 95% chance of persistence were based on data from a period of decline, that population size may be far larger than what would be required for an increasing population. While parameters estimated from a period of endangerment may provide some bench mark values, new parameters and a new MVP size would have to be estimated over the recovery period.

It is important to note that the above discussion of MVP did not include genetic considerations. Thompson (1991) concluded from his review of the literature on conservation biology that environmental variation was generally found to be the greater risk to threatened populations than was loss of genetic variation, except in populations of less than 100 individuals. We agree with that conclusion. Once a population becomes sufficiently small that inbreeding is a problem, environmental variation is likely to drive the population to extinction long before inbreeding depression becomes a problem.

Inbreeding depression is only likely to become a problem in populations of less than 100 individuals. Waples (1990) reviewed isozyme data from 177 chinook populations and

found that variant alleles most commonly occurred as rare alleles (frequency  $< 0.05$ ) (Figure 2-1). Waples used this data to simulate the rate of loss of such alleles in a salmon population with various numbers of breeders per year. Waples showed that long-term loss would be minimal if the number of breeders per year was 100, but if the number of breeders per year was 24, then about 50% of alleles with frequencies  $< 0.05$  would be lost within 100 years (Figure 2-2). However, the simulations indicated that, even over a 25 year period with only 24 breeders per year, less than 10% of the rare alleles in a population would be lost. Waples assumed that all alleles were neutrally selective. In reality, even a very small selective advantage for any rare allele would enable it to be maintained in the population, in spite of low numbers of breeders (Falconer 1981). We conclude that high variation in survival rates poses a substantially greater probability of causing extinction than does loss of genetic variation.

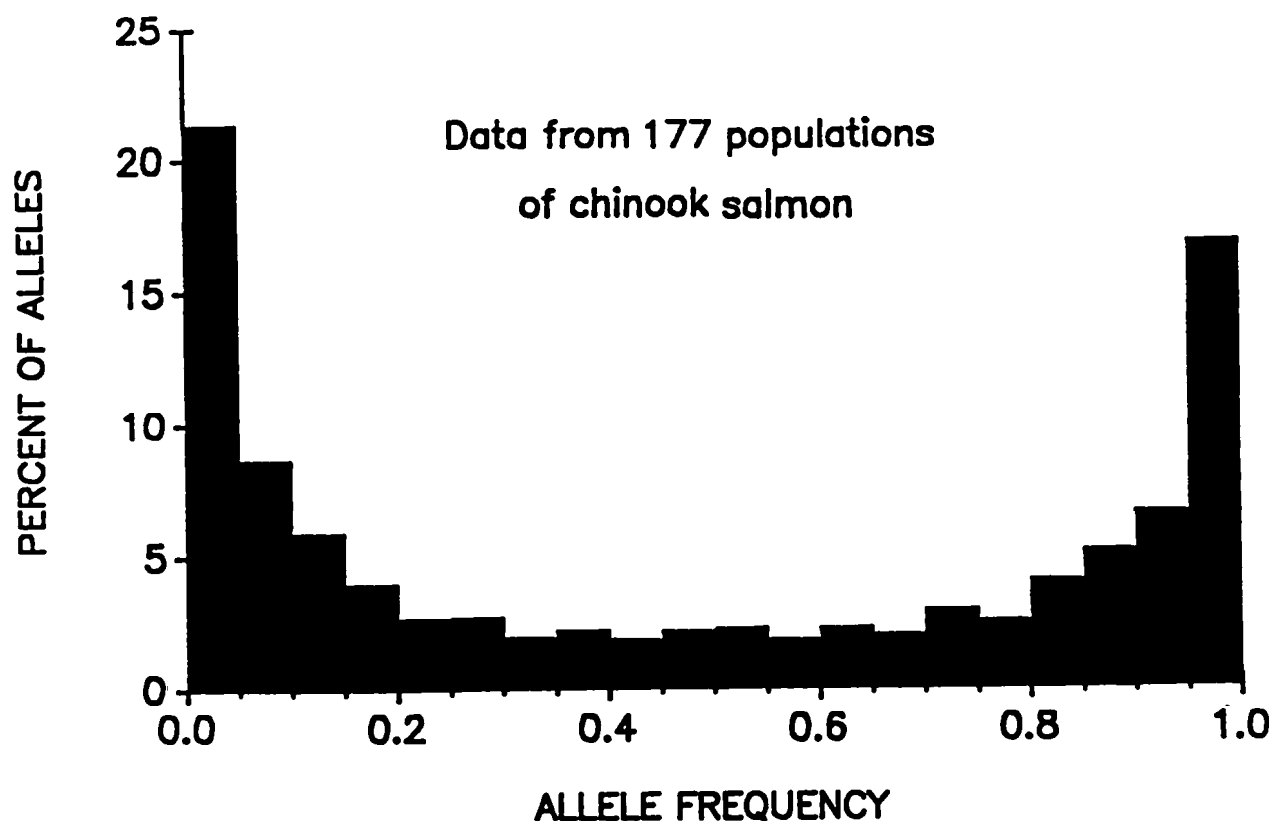


Figure 2-1. Distribution of alleles at various frequencies found in samples from 177 chinook salmon populations in the Pacific Northwest. (From Waples 1990).

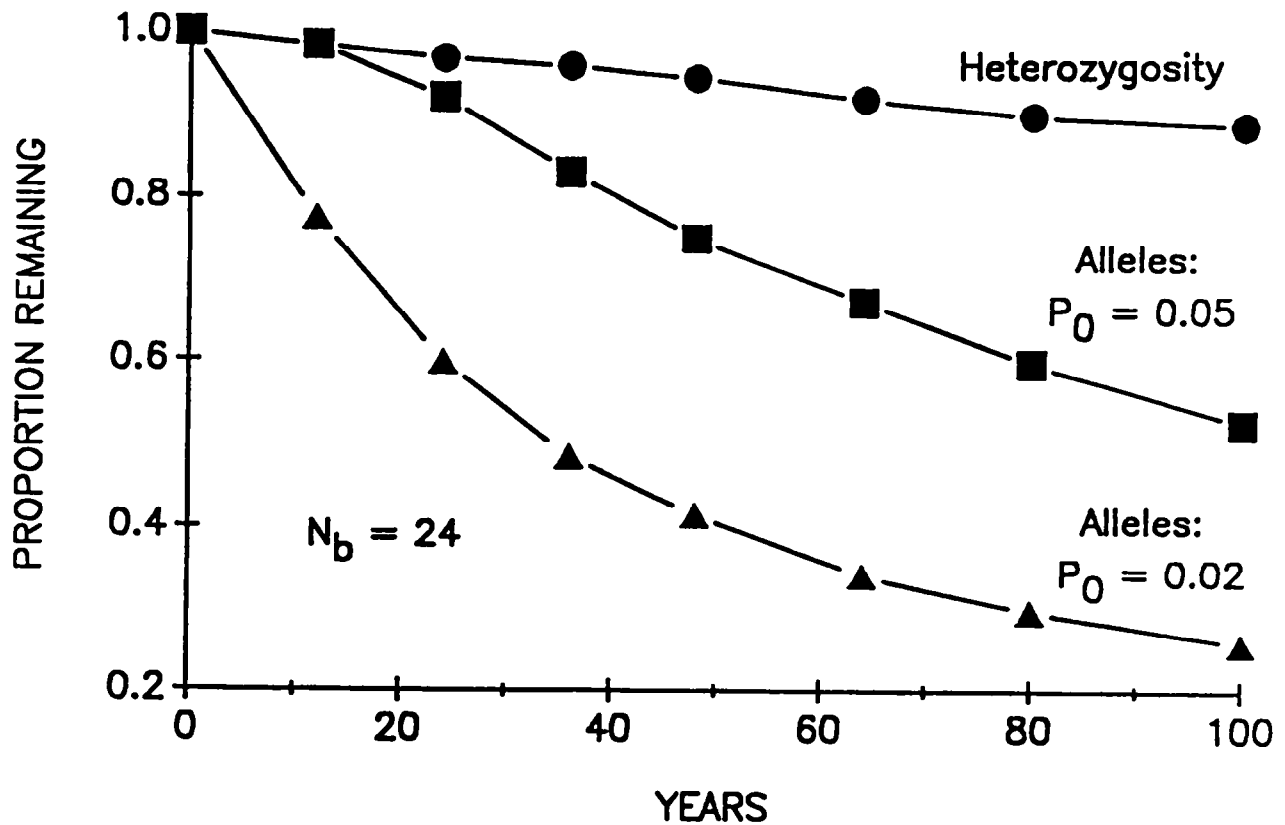


Figure 2-2. Proportion of alleles with initial frequency of 0.05 lost over time in simulated Pacific salmon populations when the annual number of effective breeders is  $N_b = 24, 50$ , and 100. (From Waples 1990).

Population size and the population time trend are not, in themselves, sufficient bases for delisting. A small population size associated with a chronic population decrease over time may provide a suitable basis for listing. However, a moderate population size associated with a reasonable period of population increase may not be a sufficient basis for de-listing. For example, fish passage improvements and increases in benign hatchery-smolt production could result in an increasing population size that reached some predetermined de-listing criterion value. However, if the spawning and rearing habit were such that the recruit to spawner ratio were less than 1.0 (recruits being natural spawners that are progeny of natural spawners in the previous generation), then the relaxation of the hatchery program would result in a return to a population decline, and the elimination of the hatchery program would ultimately result in the extinction of the stock. A stock whose long term persistence depends on artificial propagation should not be de-listed.

## 2.2 DISTINCT POPULATION SUBUNITS

If distinct population subunits exist, they need to play an important role in development of de-listing criteria, because population subunits can affect both the genetic variability of the population and the dynamics of population decline and growth. Distinct breeding subunits (the subunit is distinguished by its low gene flow from other subunits) of a population may enable maintenance of genotypes in one subunit that would be selected against in another, thereby allowing greater genetic variability to persist than if no breeding subunits existed. Distinct geographic subunits may also result in different growth, survival and maturity rates between subunits (because of differences in habitat productivity), which in turn would influence each subunits' ability to respond to an environmental perturbation, be it negative or positive. Breeding and geographic subunits may or may not correspond to each other. In the following paragraphs, we provide some specific examples of these concepts, genetic variability and population productivity, as to their importance in ESA listed Snake River salmon populations.

### 2.2.1 Genetic Variability Between Breeding Subunits

The most recent and comprehensive work on allozymic frequencies of Snake River chinook was completed by Waples, et al. (1991). Waples, et al. found a considerable increase in genetic variation at 35 gene loci over previous electrophoretic studies of Snake River chinook salmon. Based on a combined test over all gene loci, statistically significant differences in allele frequencies were found between every pair of samples, i.e. between

every population sampled. Thus, this study provided substantial evidence for restricted gene flow between streams even within the same drainage. For example, they found differences at 11 loci between fish sampled in the Secesh River versus those in Johnson Creek, which are each tributaries to the South Fork of the Salmon River. It is worth noting that the population of Secesh River is regarded as wild, whereas the population of Johnson Creek is considered to be hatchery influenced (Hassmer, IDFG, 1993, personal communication). Waples, et al., found general agreement between groupings of populations based on allele frequencies with groupings based on run timing (Figure 2-3), but in some cases found substantial differences between fish of similar run timing from different areas (e.g., between spring run samples from the Salmon and Grande Ronde Rivers). Waples, et al., concluded that both spawning time and location of spawning were important in determining genetic composition.

Genetic differences among streams inhabited by the same population could be attributed to genetic drift within streams if homing fidelity to individual streams is high and the effective number within a stream is small. Drift would be random phenomenon within streams, and different alleles would be lost in different streams. The end result would be relatively low genetic variability within streams and high genetic variability among streams. High homing fidelity may be an adaptive measure that guarantees a maximum exploitation of the habitat; once the stream has been successfully colonized, it will probably stay colonized. Limited straying among streams would guarantee that the gene pool for the whole population would be shared. This means that the genetic diversity over the habitat should be preserved.

This also suggests that there is likely to be greater genetic variability among spring-summer chinook, which exploit many tributaries over the Snake River basin, than among fall chinook, which have a far more limited spawning habitat. Genetic variability among sockeye will be severely restricted by the limited number of fish available for broodstock.



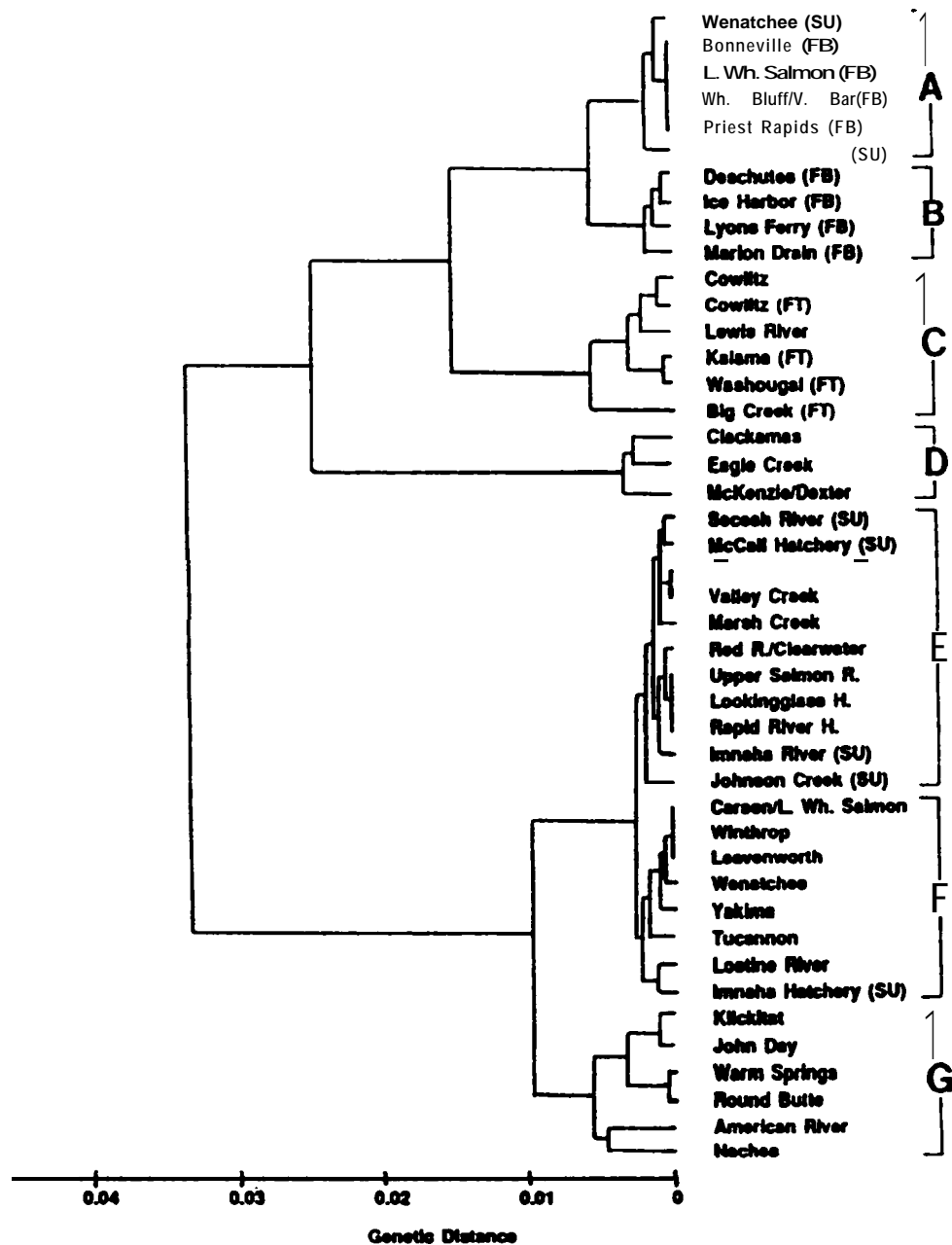


Figure 2-3. Dendrogram showing clustering of pairwise genetic distance values (Nei 1978) computed for 21 polymorphic gene loci in chinook salmon from the Columbia River Basin (from Matthews and Waples 1991).

Differences in run timing and age at maturity can also indicate genetic differences between population subunits. Time of river entry and time spawning have been found to be highly heritable traits among anadromous salmonids. Exact heritabilities have not been estimated and would differ between populations. Hager and Hopley selected coho at Cowlitz Hatchery for several different return times and found that progeny from the select groups returned predominantly in the same time period as their parents, indicating return timing was highly heritable. Garrison and Rosentreter (1981) and Ayerst (1977) demonstrated time of spawning was heritable among steelhead. Garrison and Rosentreter (1980) conducted selective breeding experiments for early and late return timing of Alsea winter steelhead and demonstrated that return timing was heritable. Garrison and Rosentreter showed that the mean date that winter steelhead returned to Alsea Hatchery gradually changed over a 30 yr period to almost 2 months earlier, and they concluded the trend was a response to unintentional selection in which the hatchery men used the earliest returning fish for brood. Ridger (1972) reviewed evidence substantiating the heritability of time of freshwater entry for chinook. Donaldson (1970) demonstrated that time of spawning was heritable among chinook.

Selective breeding experiments with fall chinook at Elk River, Oregon, demonstrated that not only is time of spawning highly heritable, but it is also correlated to time of river entry (Nicholas and Hankin 1988). In the experiments at Elk River, progeny from early spawning and late spawning fish were held separate and marked with CWTs in two consecutive brood years. The mean date of capture in a fishery at the mouth of Elk River was 6 and 11 days earlier for the two early groups than for the late groups, and the mean date of spawning for the two early groups was 19 and 26 days earlier than the mean dates of the late groups. Thus, differences between stocks in time of river entry or time of spawning are likely to reflect genetic differences between stocks.

Age at maturity is also a heritable trait. Gall et al. (1988) found that age at spawning in rainbow trout was moderately to highly heritable, and Garrison and Rosentreter (1980) were able to produce a change in average age at maturity of Alsea winter steelhead by selective breeding. Nicholas and Hankin (1988) present findings from a selective breeding experiment with Elk River fall chinook that age at maturity was highly heritable. They found that the progeny of age 3 fall chinook averaged about 1 yr younger at maturity than the progeny of age 5 parents. Downey et al. (1986) also showed that the age of the male in matings of Elk River fall chinook influenced the age at maturity of the male progeny, but not the female progeny; thus, age at maturity was a sex-linked trait.

### **2.2.2 Variation in Productivity Between Geographic Subunits**

The natal streams in which salmon spawn and rear are an example of geographic subunits. Numerous geographic population subunits (tributaries) comprise the Snake River spring/summer chinook; whereas, it could be argued that the fall chinook ESU and the sockeye ESU are each composed of only one geographic unit. The numerous geographic subunits for spring/summer chinook are pertinent to our discussion here because, IDFG (1992) has established that there is substantial variation in habitat quality both within and between the natal streams used by Snake River spring/summer chinook.

Differences in habitat quality between natal areas may affect egg-to-smolt mortality, juvenile growth rate (therefore, size at smolting), age at maturity, and prespawning survival during summer holding of adults. Each of these characteristics will, in turn, influence the rate at which the population subunit will respond to management actions. Examples of each of these effects follow. The effects of these differences on population rebuilding and on de-listing criteria is discussed further in a later section of this report on simulation modeling.

- ▶ It has been well documented that survival of salmon eggs is highly correlated the percentage of fines in the gravel (Chapman 1988).
- ▶ The average size of chinook smolts leaving the Imnaha each year is smaller than those leaving the Grande Ronde (personal communication, K. Witty, ODFV biologist, retired, Enterprise, Oregon).
- ▶ Age at maturity tends to be older for juveniles that smolt at a smaller size (Hankin 1990). Fish that mature at a later age are subjected to additional years of harvest in the ocean and, because of their larger size at maturity, are more vulnerable to the in-river gillnet fishery.
- ▶ Squawfish in the Columbia River tend to prey on the smaller sized juvenile chinook available in the river (Poe et al. 1991).
- ▶ Prespawning mortality of adults that have passed all dams can exceed 50% (Chapman et al. 1991). and this mortality can be reduced in areas with high quality holding habitat.

### 2.3 SURVIVAL RATES

Because recovery actions will be directed at removing or reducing threats to fish survival at the various stages in the life cycle, survival estimates at each major stage should be included in de-listing criteria. In order to evaluate the impact of the various recovery components, it will be necessary to measure the trends in specific life history survival rates or population numbers. The net effect of the survival rates at the various life stages is what determines whether the population is increasing or decreasing. Thus, life stage survival rates are the building blocks of the conditions that are used to estimate MVP. It follows logically that de-listing criteria based on life-stage survival rate moves one-step closer to identifying the cause of the problem and correcting it, and should provide more efficient de-listing criteria than MVP.

### 3. DESIRABLE **QUALITIES** OF DE-LISTING **CRITERIA**

Numerous population and habitat parameters have been suggested as criteria for measuring the health of salmon populations, so we began our analysis by developing a framework within which alternative criteria for de-listing could be compared. There are many qualities that it would be desirable for a de-listing criterion to have, so we have developed a conceptual checklist of these qualities against which each parameter can be rated. A description of some important qualities that we have identified follows.

- ▶ **Historic Record.** Is there a long time series of data on this variable? Historical trends cannot be analyzed if a time series is lacking.
- ▶ **Accuracy.** Is it likely that methods used in the past to measure this variable have introduced bias? Are sampling methods in the future likely to be unbiased? Some population parameters may be very difficult to measure without bias, and although the parameter might be important, the measurement may be misleading.
- ▶ **Precision.** What is the variance associated with this variable? Can precise estimates be realized? Some parameter estimates may be so variable that substantial changes in the present level would be statistically difficult to detect. In contrast, other parameter estimates may be less variable, and provide a greater opportunity for statistically detecting change.
- ▶ **Ease of Measurement.** Is this a variable that can be measured easily? It is not necessary that measurement be easy, but some weight must be given to the practicality of monitoring the variable.
- ▶ **Controllable** by Managers. Is the parameter one which can be affected directly by management actions? It is desirable that some de-listing criteria be tied directly to parameters that are intended to benefit from specific elements of the Recovery Plan.
- ▶ **Driving Variables Quantified.** Do we have any quantitative relationships to predict the causes of variation in this variable? It will be of great benefit in estimating population health if we can distinguish between variation caused by factors that we cannot change from the variation caused by factors that we can change. Without the

ability to make this distinction, environmental variation will make it difficult to distinguish the benefits achieved by the Recovery Plan.

A given characteristic may be suitable for some measures but not others. For example, redd counts from index areas may give very biased estimates of absolute abundance but may give reasonably accurate estimates of trend.

We do not develop a checklist for characteristics evaluated in this report, but certain of the qualities are evaluated within the text.

#### **4. GENERAL DISCUSSION OF PARAMETER MEASURES**

##### **4.1 ABUNDANCE MEASURES AND TREND ANALYSIS**

The population size has been used as basis of assessing health. What is crucial is

- b that the measure of population size used should reflect the effective number of naturally produced reproductive individuals (i.e, the number of wild or natural spawners),
- that the trend in spawner growth be characterized and be subjected to statistical evaluation, and
- ↗ that the trend be used to evaluate the probability of extinction

We do not evaluate the extinction probabilities in our in-depth analysis, but we do recommend specific components that should be included in any extinction model, whether it be an analytical model, like the exponential diffusion model used by NMFS in developing the decision criteria it used for listing spring/summer and fall chinook, or the application of a simulation model, like the Stochastic Life Cycle Model (SLCM, see Lee and Hyman, 1991).

Several indicator measures of spawner number should be considered:

- Redd count: This would be probably be the best measure if appropriate sampling procedures are developed and if conditions at the spawning site are conducive to counting redds.
- Number of fish reaching spawning grounds: This would not be a direct measure since all of the fish may not spawn; however this measure could still be highly correlated with the number of spawners.
- Spawn/Al-cspawned-out: carcass is one that has spent part or all of its eggs and therefore is a direct indication of spawning.

- ▶ **Parr density**: This measure would be a suitable measure if there were a reasonable estimate of the parr-to-spawner or the parr-to-redd ratio.
- ▶ **Escapement to spawning grounds**: Unless weir traps are representatively distributed over the spawning area, direct measures of escapement may not be possible. If there is a tendency of fish to avoid weir traps, then their use may affect fish distribution over the spawning habitat.
- ▶ **Natural smolt outmigration**: Counts of natural smolts may serve as an indicator of spawning ground production. Count estimates of Lower Granite Dam's bypass passage can be made.

In the remainder of this section we evaluate potential trend measures and, in some cases, touch on their application to estimating extinction probabilities. We first discuss the analytic tools used in the delisting process, we then discuss their application to the delisting process, and we finally discuss various analysis tools that can be applied to the delisting process.

#### **4.1.1 Basis of listing**

One evaluation tool that NMFS used for listing was the an exponential diffusion model to assess the probability that the population would become extinct within a given period of time (Matthews and Waples 1991 and Waples et al. 1991). Dennis (1991) derived a formula that was used to assess the extinction probability. The formula required the following information:

- ▶ the infinitesimal mean (exponential instantaneous rate of growth or decline),
- the infinitesimal variance applicable to an exponential diffusion model,
- the base population size ( $Y_{\text{base}}$ ) which is often used to establish the minimum viable population size.
- ▶ the extinction number ( $Y_{\text{ext}}$ )



Dennis also developed estimators for the infinitesimal mean and variance. These estimators utilized the log of the ratio between the abundance measures at time  $t$  and  $t-\Delta t$ ;

$$z(t) = \ln \{y(t)/y(t-\Delta t)\}$$

$y(t)$  and  $y(t-\Delta t)$  being the abundance measures. We refer to  $\Delta t$  as the lag-time, the length of time between the two measures. Dennis's probability equations and estimators are presented in Appendix 1 along with discussions relevant to the rest of this section.

**NOTE:** There was a typographical error in Dennis's extinction probability equation. The corrected form of this equation is given in Appendix 1. While we were able to duplicate the endangered probabilities of extinction presented by Matthews and Waples, we were not able to duplicate recent probabilities presented in NMFS memoranda (Scwiebe 1992, and Wainwright 1992). Nor were we able to duplicate the threatened probabilities of Matthews and Waples.

NMFS used Dennis's estimators, applying them to reconstructed Lower Granite wild fish counts for fall chinook and to redd counts for spring/summer chinook. The estimates of the infinitesimal mean and variance along with values for  $Y_{base}$  and  $Y_{est}$  were substituted into Dennis's probability formula. These extinction probability estimates were a major factor in NMFS's decision to list the Snake River spring/summer chinook and the fall chinook as threatened under the Endangered Species Act.

The following applied to NMFS estimation of the extinction probability:

- Five year-running totals: NMFS utilized five-year running totals for  $y(t)$ .
- Overlapping totals: The five-year running totals overlapped from data point to data point; i.e. the  $y$ 's were of the form

$$y(t) = c(t) + c(t+1) + c(t+2) + c(t+3) + c(t+4)$$

$$y(t+1) = c(t+1) + c(t+2) + c(t+3) + c(t+4) + c(t+5)$$

$$y(t+2) = c(t+2) + c(t+3) + c(t+4) + c(t+5) + c(t+6)$$

etc.

wherein  $c(t)$  is the count at time  $t$ . This means that the  $y$ 's shared counts; e.g.,  $y(t)$  and  $y(t+1)$  share four-fifths of their data points. This count sharing between the data points is what we refer to as "overlapping".

- Lag time = 1: The lag time used by NMFS was  $\Delta t = 1$  (Tom Wainwright, NMFS, Seattle, personal communication).
- Estimates treated as parameters: The probability of extinction within a specified time period was estimated by treating the estimated infinitesimal mean and variance as if they were the actual population parameters instead of estimates.
- Extinction number = 1: Extinction was defined as the point where  $Y_{ext} = 1$ ,  $Y_{ext}$  being a five-year running total.

#### 4.1.2 Listing Models as Applied to the De-listing Process

The analysis tools used for listing a species may not be the tools applicable for de-listing the species. Even if the same model were appropriate, it is unlikely that any of the parameters estimated from the period of decline would be applicable to the period of recovery.

Below we identify problems in NMFS's methods of applying the Dennis model. Specifically, we focus on the definition of extinction number, accommodating variability of the estimated or simulated parameter in the model, problems with using the running total instead of yearly abundance, and the choice of generational lag time. Since the problems could apply to analytical and simulation tools in general, they should be addressed whether or not the Dennis model is used to evaluate the recovery program.

**4.1.2.1. Definition of Extinction Number**

The current definition of extinction is  $Y_{\text{ext}} = 1$  individual per generation. This is far too low a number. One individual per year for each generation year would also lead to absolute extinction. Further, any number of individuals, all of the same sex within a given year for each year within a generation, would also lead to absolute extinction. Therefore, the  $Y_{\text{ext}}/\text{generation}$  should be greater than 1.

However, absolute extinction is not the only basis for assessing extinction. The return of four adult Snake River sockeye in 1991, and of one in 1992, was small enough that the decision was made to take all five into a captive broodstock program. Therefore, Snake River sockeye are currently being treated as if they were extinct in the wild. An appropriate definition of extinction in the wild should be the critical number that would force the recovery program to resort to taking all returns as broodstock.

Concern about an extinction number is not a trivial matter when the Dennis model is used. At low values of  $Y_{\text{ext}}$  increasing the extinction number can dramatically increase the probability of extinction. We use the parameter estimates of Matthews and Waples to illustrate this extreme sensitivity in Table 4-1.

Table 4-1. Probability of extinction in 100 years, $P(T < 100)$ , for infinitesimal mean ( $\mu$ ), infinitesimal variance ( $\sigma^2$ ), and base number ( $Y_{\text{base}}$ ) using estimates from Matthews and Waples (1991)		
Extinction Number $Y_{\text{est}}$	1957-1990 data set $\mu = -0.06199$ $\sigma^2 = 0.02649$ $Y_{\text{base}} = 7065$	1964-1990 data set $\mu = -0.05486$ $\sigma^2 = 0.02765$ $Y_{\text{base}} = 3720$
1	0.05	0.05
2	0.11	0.11
4	0.22	0.21
8	0.36	0.35
16	0.53	0.51
32	0.69	0.67
64	0.82	0.80
128	0.91	0.90

In this example doubling the extinction number approximately doubles the extinction probability until  $Y_{\text{est}} = 8$ .

We do not know whether the sensitivity to changing  $Y_{\text{est}}$  at low values of  $Y_{\text{est}}$  is simply a function of the analytical model used. Various analytical and simulation procedures should be evaluated and diagnosed as to their ability to project population trends and to estimate the probability of extinction within a given time interval. Part of this diagnosis should include the procedure's sensitivity to  $Y_{\text{est}}$ .

However, no matter what model is used, reasonable values of  $Y_{\text{est}}$  should be chosen. The easiest way to discuss extinction would be to focus on returns per year. We concentrate on two options, extinction based on all returns being of the same sex and

extinction based on a critical number requiring all returns to be taken as broodstock.

Extinction based all returns being the same sex. It would be possible to choose  $Y_{est}$  based on a probability of the escapement being the same sex. Assuming that the number of females is distributed binomially with an expected proportion  $q$ , the probability of having all returns being the same sex is

$$P = q^Y + (1-q)^Y$$

For an expected 1:1 sex ratio ( $q = .5$ ), the probabilities are given in Table 4-2. for specified values of  $Y_{est}$

Table 4-2. Probability of the same sex for given $Y_{est}$ when expected proportion of females is .5								
$Y_{est}$	1	2	3	4	5	6	...	10
P	1	.5	.25	.125	.0625	.03125	...	<0.005

If there is no information as to the distribution of  $Y_{est}$ , then a conservative approach would be to focus on the conditional probabilities in Table 4-2. If the decision is made to have less than a 0.1 probability of having the same sex in a given year, then  $Y_{est} = 5$  would be chosen since that is the lowest value for which the probability was less than 0.1.

If the distribution of  $Y_{est}$  were known, then less conservative approaches involving expected loss functions could be used to select an appropriate extinction number. While it would be reasonable to assume the distribution of  $Y_{est}$  to be a Poisson, we would not know ahead of time the value of the Poisson parameter. Therefore, if the return of all individuals as the same sex is to be the criterion for declaring a species extinct, then conditional probabilities for given values  $Y_{est}$  should be the focus of choosing the appropriate value of  $Y_{est}$ .

Extinction based on the decision to collect all returns for broodstock. The decision on the value of  $Y_{est}$  should be based on criteria used to judge at which point all of the population should be taken as captive broodstock. To the best of our knowledge, such criteria do not yet exist. We suggest an approach to this issue below.

In general, in order to preserve the genetic integrity of an evolutionarily significant unit within a hatchery program, the proportion of escapement to take as hatchery broodstock should be large enough to guarantee that the allelic distribution of hatchery broodstock is reasonably similar to that of the population. The minimum proportion can be chosen so as to minimize the relative difference between the broodstock's and the population's allelic frequencies. Specifically, the proportion taken as broodstock can be chosen so as to have a specified degree of certainty that the relative difference is less than a specified relative percentage; i.e., expressed as a probability

$$P\{|\hat{p} - p|/p < r\} = 1 - \alpha \text{ for } p \leq 0.5$$

wherein  $p$  is the allelic frequency in the population,  $\hat{p}$  is the allelic frequency in the sampled hatchery broodstock,  $1 - \alpha$  is the degree of certainty that the absolute value of the difference between  $\hat{p}$  and  $p$  relative to  $p$  is less than a specified value,  $r$ .

In Appendix 2 we develop the following algorithm for approximating the proportion ( $s$ ) sampled for broodstock for an allelic frequency of  $p \leq 0.5$ :

$$s = 1/[\hat{f}(N, r, p, \alpha) + 1]$$

wherein

$$\hat{f}(N, q, p, z(\alpha)) = 2N[r/z(\alpha)]^2[p/(1-p)]$$

and  $z(a)$  is the two-side, standardized, normal  $z$ -value evaluated at  $a$  and  $N$  is the population size.

In Table 4-3. we use this algorithm to assess the sampled proportion needed at different population sizes to guarantee that the relative difference between the brood-stock and population frequencies does not exceed 20% ( $r = 0.2$ ). This is done for degree of certainty probabilities,  $1 - \alpha$ , of 0.8 and 0.9 and for allelic frequencies,  $p$ , of 0.5, 0.1, and 0.05. Under Hardy-Weinberg equilibrium, 75% of the population are expected to have the allele, either homozygously or heterozygously, when  $p = 0.5$ ; the percentages are approximately 20% when  $p = 0.1$  and 10% when  $p = 0.05$ .

Table 4-3. Proportion of population (of size  $Y$ ) to take as broodstock in order to be  $100 \cdot (1 - \alpha)\%$  certain that relative difference between broodstock and population allelic frequencies ( $\hat{p}$  and  $p$ , respectively) differ by no more than 20% ( $r = 0.2$ ); i.e.

$$P\{ |\hat{p} - p| / p < r \} = 1 - \alpha$$

Popula- tion Size ( $Y$ )	$1 - \alpha = .8$ $p = .5$	$1 - \alpha = .8$ $p = .1$	$1 - \alpha = .8$ $p = .05$	$1 - \alpha = .9$ $p = .5$	$1 - \alpha = .9$ $p = .1$	$1 - \alpha = .9$ $p = 0.5$
12800	0.002	0.014	0.030	0.003	0.023	0.048
6400	0.003	0.028	0.057	0.005	0.045	0.091
3200	0.006	0.055	0.109	0.010	0.087	0.167
1600	0.013	0.104	0.196	0.021	0.160	0.287
800	0.025	0.188	0.328	0.041	0.276	0.445
400	0.049	0.316	0.494	0.078	0.432	0.616
200	0.093	0.480	0.661	0.145	0.604	0.763
100	0.170	0.649	0.796	0.253	0.753	0.865
90	0.186	0.673	0.813	0.273	0.772	0.877
80	0.204	0.698	0.830	0.297	0.792	0.889
70	0.227	0.725	0.848	0.326	0.813	0.902
60	0.255	0.755	0.867	0.361	0.835	0.915
50	0.291	0.787	0.886	0.404	0.859	0.928
40	0.339	0.822	0.907	0.458	0.884	0.941
30	0.406	0.860	0.929	0.530	0.910	0.955
20	0.507	0.902	0.951	0.628	0.938	0.970
10	0.673	0.949	0.975	0.772	0.968	0.985

Focusing on  $p = 0.1$  and  $a = 0.9$  in the table, more than 90% of the population would be sampled if the population size were 30. This might correspond to a point where taking the whole population for broodstock might be considered. A larger number may be justified since the normal approximation to the binomial distribution would be breaking down at  $N = 30$  for  $p = 0.1$ .

Naturally, the whole population would be taken as broodstock only if the expected return from a captive broodstock program was deemed to be higher than that from natural spawners under low population sizes.

The focus of the above discussion has been spawner count per year as opposed to spawner count per generation. Extinction applied to generation count would have to incorporate assumed probability distributions of returns over years within a generation.

#### 4.1.2.2 Accommodating the Variability of Parameter Estimates in the Model

Dennis's extinction probabilities are based on the parameter,  $\mu$ , not on the parameter estimate. There are two sources of variation that would contribute to the estimated extinction time,

- inherent year-to-year (or generation-to-generation) variation in population growth or decay, and
- variation in the parameter estimate itself.

This is not a trivial issue. The variation in the estimate can have a measurable impact on the estimated probability of extinction. This is because an incremental decrease in the mean (i.e. increase in rate of decline or decrease in the growth rate) will increase the extinction probability more than an equivalent incremental increase in the mean will decrease the extinction probability (Dan Goodman, Department of Biology, Montana State University, 1993, personal communication).

It is possible to accommodate both sources of variation in estimating the expected extinction probability for the Dennis model. If the data variables used to estimate the infinitesimal mean are independent, then the variance of the estimated infinitesimal mean can be easily estimated (Appendix 1). The estimate's distribution can be approximated by substituting the estimated infinitesimal mean and variance for the population infinitesimal



mean and variance in the normal probability density function. The probability of extinction for a given estimated infinitesimal mean can be integrated over the probability density function for the estimate. Exact forms of this integral may not exist; however numerical techniques can be used to approximate the integral. We have done so using the parameter estimates of Matthews and Waples and using  $At = 1$  giving  $n = 33$  for the 1957-1990 data set and  $n = 26$  for the 1964-1990 redd count data that they used. The results are summarized in Table 4-4.

<b>Table 4-4. Effect of accounting for distribution of mean estimate on estimate of probability of extinction in 100 years, <math>P(T &lt; 100)</math>, for infinitesimal mean (<math>\mu</math>), infinitesimal variance (<math>\sigma^2</math>), and base number (<math>Y_{base}</math>) using estimates from Matthews and Waples (1991)</b>		
Effect of distribution of $\mu$ taken into account?	1957-1990 data set $\mu = -0.06199$ $\sigma^2 = 0.02649$ $n = 33$ $At = 1$ $Y_{base} = 7065$	1964-1990 data set $\mu = -0.05486$ $\sigma^2 = 0.02765$ $n = 26$ $At = 1$ $Y_{base} = 3720$
no	0.05	0.05
yes	0.21	0.18

There was an approximately four-fold increase in the probability of extinction due to variation in the estimated infinitesimal variance. We did not investigate the effect of variation of the estimate of infinitesimal variance on the probability of extinction.

#### 4.1.2.3. Problems with the Use of Running Totals to Estimate Trend Parameters

There are two major reasons that the running totals/averages should not be used. The first relates to bias of the estimate of the infinitesimal mean and the second relates to the bias of the estimate of the infinitesimal variance.

**Bias in estimating infinitesimal mean:** The log of the ratios of the running 5-year totals,

$$\ln\left\{\frac{c(t) + c(t+1) + \dots + c(t+4)}{c(t-\Delta t) + c(t-\Delta t+1) + \dots + c(t-\Delta t+4)}\right\}$$

when divided by  $\Delta t$  will give a biased estimate of the infinitesimal mean.

We note that using the generation average per year,

$$y(t) = [c(t) + c(t+1) + \dots + c(t+5)]/5$$

would produce the same result as the running total since the number of years (5) in  $\ln\{y(t)/y(t-\Delta t)\}$  would cancel; therefore, we use the terms running total and running average interchangeably.

**NOTE:** The interchangeable running total and running average complicates our previous discussion on extinction number. We are not certain whether the extinction number is a generational extinction number (running total) or a yearly extinction number (running average).

If it is deemed appropriate to use generation summaries as the data base for assessing trend, then the geometric mean

$$y(t) = [c(t) * c(t+1) * \dots * c(t+4)]^{1/5}$$

should be considered since  $\ln\{y(t)/y(t-\Delta t)\}/\Delta t$  using this measure would be an unbiased estimate of  $\mu$ .

**The running totals/averages and geometric means will give biased estimates of the infinitesimal variance:** The estimated infinitesimal variance will be smaller, possibly much smaller, than it should be because the estimates share common data points, reducing the variation between them. Such bias would also exist for the estimated variance based on overlapping 5-year products. The variance among independent means is expected to be less than that among independent observations; the bias that we are discussing results in an even smaller variance because the means are positively correlated due to their sharing

common data points.

In Table 4-5, we give different estimates of the infinitesimal mean and variance from the 1975-1992 wild fall chinook Lower Granite dam counts (Appendix 3). The estimates in the table are based on yearly log-ratios, log-ratios of 5-year running totals/averages, and log-ratios of 5-year geometric means.

Table 4-5. Estimates of infinitesimal mean ( $\mu$ ) and variance ( $\sigma^2$ ) for wild fall chinook Lower Granite Dam counts (1975-1992) based on logs of the ratios of yearly counts, 5-year running totals/averages, and 5-year geometric means for different lag times				
Lag time $\Delta t$	Parameter estimate	Yearly counts	5-year running totals/averages	5-year geometric means
1	$\mu$	-0.0370	-0.0539	-0.0637
2	$\mu$	-0.0319	-0.0587	-0.0686
3	$\mu$	-0.0680	-0.0597	-0.07059
4	$\mu$	-0.0685	-0.0589	-0.0676
5	$\mu$	-0.0637	-0.0587	-0.0662
1	$\sigma^2$	0.3933	0.01359	0.01554
2	$\sigma^2$	0.2695	0.00864	0.01182
3	$\sigma^2$	0.0589	0.00263	0.00469
4	$\sigma^2$	0.0748	0.00355	0.00575
5	$\sigma^2$	0.0777	0.00497	0.00655

As expected, the variances for both the 5-year running totals/means and the 5-year geometric means are considerably smaller than those based on yearly log-ratios. Since the

variance of a mean estimate is the population variance divided by the sample size if the sample elements are independent, variances based on individual log-ratios are expected to be greater than the variances based on the geometric mean by a factor of the number of data points used to compute those means. However, the magnitude of the differences in Table 4-5. is far greater than that number, 5; the factor ranging from approximately 12 to 25. The fact that the estimated variance among geometric means is far smaller than would be expected may be due to the bias from using overlapping data. However, analysis of the 1964-1990 total spring/summer chinook redd counts used by Matthews and Waples (data summaries not presented) was not consistent in this regard. The estimated variance among geometric means was lower than expected for the shorter lag-times, as was the case in Table 4-5 for wild fall chinook dam counts; however, the variance was higher than expected for the longer lag-times.

The yearly log-ratios are unlikely to be independent. They are probably positively correlated due to similar climatic conditions in adjacent years and due to the fact that returns in adjacent years share brood years.

The geometric mean estimates a greater loss than does the running total. This was also true for estimates that we generated from the 1964-1990 total spring/summer chinook redd counts used by Matthews and Waples. It appears that the estimated trends based on running totals give a smaller loss estimate than those based on the presumably unbiased geometric means and that NMFS may be underestimating the extinction probabilities of the listed stocks.

In Table 4-5. the geometric mean and running total estimates are far less variable over lag times than are yearly log-ratio estimates.

(Note: The decrease in the yearly log-ratio estimates with lag time in Table 4-5 is only a function of the data set. The same analysis was performed on the 1964-1990 redd count data used by Matthews and Waples, and there was no such trend.)

#### **4.1.2.4 Establishing Appropriate Lag Time**

The lag time should correspond to the effective generation length. Given that the age composition of the return reflects different brood-year contributions, the lag time should probably be weighted averages of the age of the returning (r) spawners, the weights being the age distribution of the returning fish. These same weights should be applied to the

lagged spawner number of the previous parental (p) generation; e.g., the yearly ratio used in trend analysis for age 4 and 5 spring/summer adult returns should take the form:

$$c_r(t)/c_p(t-\Delta t)$$

wherein

$$c_r(t) = c(t)$$

$$\Delta t = w(4)*4 + w(5)*5$$

and

$$c_p(t-\Delta t) = w(4)*c(t-4) + w(5)*c(t-5)$$

$w(4)$  and  $w(5)$  being the age distribution of the spawners, and  $c_p(t-\Delta t)$  being the weighted counts of the contributing brood years.

In all our analyses we used  $\Delta t=5$  for simplicity; however, weights should be developed for each population with different age distributions.

**RECOMMENDATION: Any model used to assess extinction probabilities should accommodate both the variation in the estimated or simulated parameter and random variation. The model should be checked for its sensitivity to extinction number and should use lag-times that reflect the generation length.**

#### 4.1.3 **Models Applied to the Recovery Period and to Recovery Goals**

Various models, including both analytic and simulation models, should be investigated. Any models investigated for the purpose of developing de-listing criteria should:

- ▶ be able to estimate the number of wild spawners and be able to use those estimates to test hypotheses regarding the wild spawner population
- ▶ be able to estimate the rate of growth or decline in wild spawner number over time and be able to use those estimates to test hypotheses regarding the population trend,
- ▶ be able to estimate the probability of extinction given the spawner number and trend,

or

- be able to provide information on criterion de-listing target values for the parameters being estimated.

Statistical model selection for a given assessment need should be based on the following criteria

- a) Are the model parameters easily understood?
- b) Is the model applicable?
- c) Is it easy to fit the data to the model?
- d) Can the variability in the parameter estimates be characterized?
- e) Can the parameters be easily tested statistically?
- f) What are the relative biases and precisions of the parameter estimates?
- g) Is it possible to adjust for covariates that are
  - a) not functions of the recover program, and
  - b) likely to affect abundance or trend?
- h) Is it possible to characterize "random" variability for the purpose of evaluating extinction probabilities?

We discuss these criteria in conjunction with the specific model estimators, but we expand here on criteria g) and h).

Regarding g), the ability to adjust for covariates that are unrelated to the recover program and are likely to affect abundance or trend: Drought years is an example of such a factor. What we mean by "not functions of the recovery program" is that the recovery

program did not cause the variability in this driving factor. Although the recovery program may take action to mitigate against the effects of drought, the recovery program will not stop a drought from occurring and will probably not be able to fully compensate for the effects of drought on fish abundance or trend. It may be appropriate to adjust for such factors in some models but not in others. For example, for the purpose of obtaining precise estimates of trend over the recovery program, adjusting for drought may be appropriate; however for the purpose of estimating extinction probabilities it may be more appropriate to treat drought as a source of "random" variability that would tend to increase the probability of extinction. (Drought occurrence is probably not a random factor. It is unlikely that the 6-year drought of 1987-1992 was a series of random events.)

Regarding h), characterizing "random" variability for the purpose of evaluating extinction probabilities, this measure is specifically directed at extinction probabilities. In general, the random variability measure would have to be accommodated in the statistics tests under d), although the accommodation of that random variability in statistical tests is not straight-forward for most non-linear models.

We concentrate on two statistical models that can be used to summarize monitored data in terms of trend and abundance: the exponential model and the Ricker function.

#### **4.1.3.1 Exponential Trend Models**

The exponential growth/decay model should be the major analytical tool in assessing the early part of the recovery program because the population should be experiencing an increasing growth rate.

Exponential trend models are of the form

$$y(t) = A \cdot \exp(B \cdot t) \cdot e$$

in which the spawner abundance measure ( $y$ ) at time  $t$ , is a function of time, the abundance ( $A$ ) at time  $t = 0$ , the instantaneous rate ( $B$ ) of growth or decline, and  $e$  is a multiplicative error, "exp" being the exponential constant.

- a) Are the model parameters easily understood? Instantaneous exponential growth ( $B > 0$ ) or decline ( $0 < B$ ) is a measure on a per-unit-time basis (e.g., per year) that would apply if the population growth or decline could be evaluated continuously.

This is usually not the case, salmon abundance is usually measured within a specific period of time; e.g., redd count during the spawning season or dam count during run time. However, the following can be used to translate the estimate of B into the estimate of the actual proportion growth or decline:

$$P = 1 - \exp(B) \text{ for } B < 0 \text{ (decline)}$$

or

$$P = \exp(B) - 1 \text{ for } B > 0 \text{ (growth)}$$

wherein P is the proportion growth or decay on a yearly basis. For example, for  $B = -0.1$  (an instantaneous measure of 10% decline in population),

$$P = 1 - \exp(-.1) = 0.095$$

(a 9.5% yearly decline), or for  $B = 0.1$  (an instantaneous measure of 10% growth in population),

$$P = \exp(0.1) - 1 = .105$$

(a 10.5% yearly increase).

For reasonably small values of B, the instantaneous measure is a reasonable approximation to the yearly change (the instantaneous  $B = -.1$  or instantaneous 10% decline does not differ greatly from the yearly 9.5% decline, and  $B = .1$  or 10% instantaneous growth does not differ dramatically from the yearly 10.5% increase yearly growth). However, the above computations are advised for meaningful interpretation.

- b) Is the model applicable? The model may be applicable to time periods of decelerating decline (the historical record) or accelerated growth (the recovery period). At a point where the spawner density of the population is high enough to begin to reduce the production per spawner, this model will no longer be appropriate, and other models, such as the exponential model, the Beaverton-Holt model or the Ricker model (the latter to be discussed in the next section), would have to be considered. However, the exponential model would probably be an effective model during most of the recovery-assessment period.



There are several methods of estimating the parameters, three of which are briefly discussed below and discussed in terms of the other criteria c) through h) in Appendix 4.

#### Simple linear regression (SLR) of $\ln(y)$ on $t$ .

The natural log transformation of the exponential model gives

$$\ln(y) = \ln(A) + B \cdot t + e$$

wherein  $e = \ln(e)$

A simple linear regression of  $\ln(y)$  on  $t$  will provide estimates of  $A$  and  $B$ . As can be seen from Appendix 4, the technique meets all of the above criteria except for two: The estimate of  $A$  is biased, and can sometimes be quite biased; and the model parameters cannot be fit to the data when the observed abundance measure,  $y$ , is zero without some modification of the data that is likely to produce biased estimates of  $B$ . The biased estimate of  $A$  frequently results in poor fits when retransformed to the abundance number.

Compute the mean of the log of ratios (MLR): This is the same estimate that Dennis uses for assessing extinction probabilities (Appendix 1). The estimate is of the form

$$b = \text{mean}(z) / \Delta t$$

wherein

$$z(t) = \ln[(y(t)/(y(t - \Delta t))]$$

$\text{mean}(z)$  being the mean of the  $z(t)$ 's over time. This is strictly a trend measure. It cannot be used to predict  $A$  and, therefore, cannot directly be used to predict abundance; it can be used to predict abundance if the previous generation's abundance is specified. This estimate is expected to be a less precise measure than that from the SLR. Like the SLR, the data will have to be modified if zeros are present.

Poisson regression of  $y$  on  $t$ . This is a generalized linear regression technique that does not require the log transformation of data. The parameters are not fit using the traditional least squares technique, rather they are fit using maximum likelihood techniques. The method is applied directly to the multiplicative model under the assumption that the

abundance at any given time has a Poisson distribution, which is a reasonable assumption for the low survival probabilities of an individual surviving from an egg to a returning spawner. The technique is expected to give unbiased, or reasonably unbiased, estimates of both A and B. It can be fitted when the observed abundance is zero. However, the appropriate statistical techniques for testing or predicting are not familiar to most biologists.

Need to accommodate biological criteria as well as statistical criteria. In appendix 2 we assess these estimators for precision and bias using the redd count data for spring chinook. These three trend estimators should also be investigated during the recovery program. However, statistical criteria should not be the only basis of evaluation. In the assessment, the log-ratio estimate was the least precise estimate of the three, which was expected to be the case; however, of the three estimators, the log-ratio is biologically the most meaningful. The estimator, when properly applied, is not based on a direct relation of abundance to time, it is based on direct relation of abundance in one generation to that in the previous generation. The log-ratio should always be evaluated even if statistical decisions are to be based on other estimates. The log-ratio also forms a base against which an estimate of one the Ricker function parameters can be tested (Appendix 4).

RECOMMENDATION: The various methods of estimating the exponential rate of growth for the recovery period and of decay for the eight-dam historic era should be investigated and used to evaluate population growth during early part of the recovery program.

#### **4.1.3.2 Density Dependent Models**

There will be a point at which the growth rate will begin to slow down. This corresponds to a point at which competition begins to set in. The growth rate will continue to decrease until the carrying capacity or replacement point is reached. The carrying capacity could be exceeded, but if it is, the population would decline back to that carrying capacity. There are several models that include carrying capacity as a parameter: the logistic, the Beverton-Holt, and the Ricker functions are among them. We focus on the Ricker because it explicitly parameterizes the point at which the growth rate begins to slow down.

The Ricker function (Ricker 1975) is a model that incorporates density dependence into the relation between spawner recruit and brood-year spawner. For our purpose, where we concentrate on returning spawners, the function takes the form

$$y(t) = y(t-\Delta t) \cdot \alpha \cdot \exp[-\beta \cdot y(t-\Delta t)] \cdot e$$

$$= y(t-\Delta t) \cdot \exp[a - \beta \cdot y(t-\Delta t)] \cdot e$$

wherein  $a = \ln(\alpha)$ . The model is usually fit using the transformed form

$$z(t) = \ln[y(t)/y(t-\Delta t)] = a - \beta \cdot y(t-\Delta t)$$

- a) Are the model parameters easily understood? The rate of growth is expected to increase until the population size reaches  $1/\beta$  (maximum production), at which point the growth rate begins to decrease. However the population continues to grow until  $\ln[y(t)/y(t-\Delta t)] = 0$ . This is the equilibrium point or the replacement value where the recruit spawner number equals the parent spawner number. At this point the population size is  $\ln(\alpha)/\beta$ .
- b) Is the model applicable? The model is applicable to a stable environment; i.e., it is applicable to an environment that is not degrading over time. For a changing environment, the parameters would themselves be changing; and it would not be possible to know what was being estimated in the presence of parameter change unless the change in the parameters are themselves parameterized. Therefore, the Ricker function should not be applied directly to the historic record that involves a changing number of dams. It may be applicable to the eight-dam era. Ricker function estimates applied to the historic eight-dam era would indicate that the carrying capacity has already been reached. The population is not currently healthy, and the Ricker function shouldn't be applied as a recovery assessment tool until there is a strong indication of growth.

In Appendix 4., where we discuss statistical issues, it is mentioned that the Ricker parameter  $\beta$  will not be precisely estimated during the early stages of the recovery program. However, that parameter is a very meaningful one. The Ricker function is discussed in greater detail in Chapter 5.

**RECOMMENDATION:** The Ricker function should be evaluated in later stages of the recovery program and the population size should be tested as to whether the point of maximum growth,  $1/\beta$ , is characteristic of a healthy population.

**4.1.3.3 Trend Assessment** can be Used to **Estimate** Abundance

We consider the geometric mean of population size to be the appropriate measure for abundance during periods of exponential growth or decay. The estimate can be understood by considering a population in exponential decline. For a population declining by 50 percent a year beginning with an initial population of 512, the expected number for the first nine generations is given in Table 4-6.

Table 4-6. Expected abundance under 50% decline per year with initial abundance of 512									
Generation	1	2	3	4	5	6	7	8	9
Abundance	512	256	128	64	32	16	8	4	2

The simple average of these abundances is 114 which is roughly comparable to the population size in the 3rd generation, or near the beginning of the decline; the simple average will be skewed toward the higher values for populations in exponential decline or growth. The geometric mean is 32, exactly at the generational mid-point. The geometric mean estimates the abundance halfway through an evaluated exponential growth or decay process.

The mean can be estimated by simply estimating the mean of the logs of the transformed data

$$\ln_1(y) = \text{mean}\{\ln(y)\}$$

and by then retransforming that mean:

$$\bar{y} = \exp[\ln_1(y)].$$

Statistical tests applied to the means of log-counts often are more powerful than tests applied to the arithmetic means when applied to counts. This is illustrated using the total of redd count data over twenty-three index areas that were enumerated in each year from 1978 through 1992. (The South Fork Salmon River index area was excluded for reasons given in the next section, and the specific index areas that were used are listed in

a table in Appendix 4.) When we apply a t-test to the mean of the counts under a hypothesized value of one (the extinction value used by NMFS), the value of the t is 3.88. However, if we apply the t-test to the mean of the logs under a hypothesized value of zero (a zero log corresponds to the hypothesized count of one), then the t-value is equal to 40.46. The capability of detecting a significant difference can be greatly enhanced by using the log transformation when it is appropriate.

The standard error for the mean will often be larger than it should be because no adjustment has been made for time or driving variables. Prediction equations obtained from the exponential regression tools mentioned earlier can be used to estimate the abundance at the mean time and to produce a standard error that is adjusted for the time trend.

#### **4.1.3.5 Statistical assessments applied to NMFS's no-jeopardy goal**

NMFS's no-jeopardy goal is to demonstrate "with reasonable certainty that the average number of spawning adults expected during the last of four life cycles (2005-2008) will be equal to or greater than the number observed during the 1986-1990 base period if the improvements implemented in 1993 are continued through time" (as quoted by Wainwright, 1992). We will refer to the period of evaluation (1993-2008) as the recovery period.

We are concerned about the use of the historical 1986-1990 record as the target for recovery. There is a distinction between a base and a target. The base would be what existed before the beginning of the recovery period. A target is the goal of the recovery. The goal should be well above that of the base. The 1986-1990 period included some of the lowest reconstructed dam counts and redd counts in history (Appendices 3, 5, and 6). The goal's base is dangerously low, and using it as a target could further endanger the species, irrespective of any conclusions derived from any modeling effort.

Even using the 1986-1990 record as a base for comparison can be dangerous to the fish unless adjustments for climatic conditions are made. The 1989 and 1990 returns will consist of brood years whose outmigration would have been affected by the 1987-1992 drought. If climatic conditions through the recovery period are more favorable to spawner production than the conditions affecting the 1986-1990 escapement, then population gains may be realized that may have had nothing to do with the recovery program. If realized gains are solely due to improved climatic conditions, judging the recovery program a success based on those measured gains could be catastrophic to the stock. A prolonged

drought period following the recovery evaluation period could bring the population below the 1987-1990 return-year level resulting in a crash and possible extinction. Population sizes attained in the recovery period should be high enough to guarantee that extinction is unlikely even when climatic conditions are extremely poor.

**RECOMMENDATION: Use the whole of the pre-recovery eight-dam era as a base and adjust the abundance for a drought covariate.**

Rather than focus on the last generation of the pre-recovery period, we recommend using all years for which a brood-year would have encountered eight-dams. For stock which have a high proportion of age-five spawners and which tend to out-migrate at age two, 1978 would be the first return year of the eight dam era. Defining the eight-dam era as 1978-1992 for age-five returns gives a three-generation base of comparison.

A covariate for drought, which impacts out-migrating smolt, would best be applied to the out-migration year. A covariate for something like an el Nino event which impacts ocean survival would be applied to the return year. Such covariates can be used to adjust the historic base estimate to a level that corresponds to conditions that characterize an evaluated segment of the recovery period. Table 4-7 can serve as guide to constructing covariate indicators.

**Table 4-7. Years\* associated with el Nino events and droughts from over the eight dam era**

el Nino event years*		Drought years
(1976/77/78)		1979
1982/1983	(1982/83)	1981
	(1986/87)	1985
	(1991/92/93)	1987-92

- Numbers outside parentheses, NMFS (Factors for decline, a supplement to the notice of determination for Snake River fall chinook salmon under the endangered species act, 1991);

Numbers inside parentheses, Gerald Bell, personal communication, (NOAH, Climate Analysis Center, Washington, D.C., 1993)

When applied to return abundance, a covariate index for drought could take on the following values for stock comprised primarily of age 4 and 5 fish.

$x = 1$  if both age 4 and age 5 returns out-migrated during drought years

$x = \frac{1}{2}$  if only one or the other of the age 4 and age 5 returns out-migrated during drought year

$x = 0$  if neither of the age 4 or age 5 returns out-migrated during a drought year.

#### 4.2 HERITABLE DEMOGRAPHIC CHARACTERISTICS

As mentioned in an earlier section, certain demographic characteristics in some chinook stocks are known to be heritable. We later investigate such characteristics that appear to vary over the spawning habitat. These characteristics could serve as potential criteria for identifying subunits to be considered individually when evaluating recovery.

#### 4.3 SPAWNER-TO-RECRUIT RATIOS AND **SMOLT-TO-ADULT SURVIVALS**

Broad assessments of major life cycle contributions to survival from one generation to the next can be assessed by measuring

1. spawner-to-recruit ratios where the recruit is defined as a smolt
2. smolt-to-adult ratio

Such ratios can probably be best measured at Lower Granite dam for spring/summer chinook and perhaps for fall chinook if adjustments for straying of out-of-basin fish can be adjusted for. However, in the case of Snake River fall chinook some reservoir spawning in tailrace areas may occur in Snake River reservoirs below the Lower Granite Dam.

The spawner-to-recruit ratio should be based on only wild smolt, and the smolt-to-adult survival ratio should only be based on wild fish. The latter measure would require that scales be taken from sampled wild adults to permit them being allocated to the appropriate out-migrant year.

If these measured are to be used, all Columbia and Snake River hatchery salmon will have to be marked by a clearly visible mass mark. The Pacific States Marine Fisheries Commission's (PSMFC) Subcommittee on Mass Marking (1993) recommendation that the clipped adipose fin (ad-clip) not be de-sequestered for use with the CWT should be rejected. Such a de-sequester would probably be necessary if the ad-clip were to be used to mark all Columbia and Snake River hatchery salmon. There is probably no mark comparable to the ad-clip that can be used for mass marking and subsequent mass identification. The PSMFC Subcommittee suggests the ventral fin clip be used as a mass mark for all hatchery fish; however the subcommittee also indicates that this mark is only semi-permanent, regeneration does occur. Even if ventral fin regeneration problem can be solved, this mark is unlikely to be as easily read from viewing windows or from video imagery as is the ad-clip.



## 5. DE-LISTING CRITERIA FOR SPRING/SUMMER CHINOOK

## 5.1 STOCK-RECRUITMENT RELATIONSHIPS AND CARRYING CAPACITY

**5.1.1 Ricker Function**

The Ricker stock-recruitment function describes the mathematical relationship between the abundance of spawners in the parent generation and the abundance of their offspring that are recruited to the next generation. This mathematical function is designed to reflect the density-dependent mortality that occurs between the time that the parents spawn and the time their offspring reach the defined level of recruitment. Density-dependent mortality means that the rate of mortality increases as density increases. This increase in mortality may result from competition, attraction of predators, disease, etc. The Ricker stock-recruitment equation is:

$$R = \alpha P \exp(-\beta P) \quad (1)$$

where "exp" is the exponential constant and

R = Number of Recruits

P = Number of Parents

$\alpha$  = Parameter defining maximum value of R/P

$\beta$  = Parameter defining maximum value of R

The above equation can be converted to a linear form by taking the logarithm of both sides and rearranging the equation to:

$$\ln(R/P) = \ln(\alpha) - \beta P \quad (2)$$

The parameters of the function are usually estimated from equation (2) by least squares regression of  $\ln(R/P)$  on P. Alternatively, the equation can be expressed as:

$$R = P \exp[\alpha(1 - P/P_m)] \quad (3)$$

where,

$$a = \ln(\alpha)$$

$P_r$  = Number of parents at the level of replacement (the level where  $R = P$ )

We introduce this additional form of the function, because some workers who have estimated the parameters of the Ricker function have reported their parameter values in this form. Additionally, this form is useful for estimating the value of  $\beta$  for each stream, based on the estimated smolt capacity of that stream. Smolt capacity can be converted to the maximum achievable number of recruits,  $R_m$ , by multiplying smolts by the expected smolt-to-adult survival rate. The following is how  $\beta$  can be estimated.

Ricker (1975) demonstrated that the maximum number of recruits,  $R_m$ , is given by:

$$R_m = [\exp(a-1)](P_r)/a \quad (4)$$

If we can obtain an independent estimate of  $\alpha$  from a comparable population, then we can substitute  $a = \ln(\alpha)$  into equation (4) and we can substitute the estimated smolt carrying capacity (converted to adult recruits) for  $R_m$  in equation (4), so that we can solve for  $P_r$ . Once we have solved for  $P_r$ , it can be shown that,

$$\beta = a/P_r$$

so we can solve for  $\beta$ .

The values of  $\alpha$  and  $\beta$  reflect the units of measure applied to  $P$  and  $R$ . This becomes obvious when one realizes that  $a = R/P$  at small stock sizes. If  $R$  is expressed in terms of smolts,  $\alpha$  will be substantially greater than if  $\alpha$  is expressed in terms of adults in the ocean. Since  $\alpha$  is linearly related to  $R/P$ , any change in  $R/P$  will be reflected by a proportional change in  $\alpha$ . It is important to understand this principle, because comparison of  $\alpha$  values reported from studies of various stocks can be confused if the values used for either Parents or Recruits in the different studies represents even a slightly different life stage. For example, the value of  $\alpha$  will be smaller if Parents is taken as the spawning escapement before prespawning mortality than if Parents is the number of fish surviving to successfully spawn. Rarely do different studies that estimate the Ricker parameters use precisely the same measures of Parents and Recruits; so, before  $\alpha$  parameters are compared, the data must be adjusted to life stages that are equivalent between studies.

### 5.1.2 Review of Literature Values of $\alpha$

We evaluated the consistency of reported values for the Ricker  $\alpha$  parameter that have been estimated for spring chinook populations in the Columbia Basin. Chapman et al. (1982) estimated the Ricker parameters for the population of all spring chinook in the Columbia Basin above Bonneville Dam during the period 1938-1946, which precedes construction of all mainstem dams that fish could pass, except Bonneville Dam and Rocky Reach Dam. In a later paper, Chapman (1988) cited his 1982 study and concluded from his review of stock-recruitment functions for spring chinook that 68% is the best estimate of harvest rate that would have achieved MSY (minimum sustainable yield) for spring chinook in the Columbia Basin before dams were in place. Similarly, Lindsay et al. (1989) estimated from empirical data on spring chinook in the Warm Springs River that 75% was the harvest rate that would achieve MSY. Lindsay et al. (1985) estimated that the harvest rate for MSY of spring chinook in the John Day River was 65%, but was 75% for the population in Granite Creek, a tributary to the John Day River. Reisenbichler (1990) estimated Ricker stock-recruitment parameters for several Columbia River populations of spring chinook in recent years, but we chose not to use these because they were confounded by the effects of many mainstem dams.

### 5.1.3 Adjustment of Ricker $\alpha$ Parameter for Equivalency

The studies just cited appeared to be producing similar estimates of  $\alpha$  for different populations of spring chinook within the Columbia Basin, and we desired to identify a reasonable value of  $\alpha$  for the Snake River population, so we carefully examined the measures of Parents and Recruits used in each of the studies cited. Additionally, we wanted these values to be equivalent to what would have occurred before dams were built in the river. It was important for our purpose to use the pre-dam stock-recruitment function so that we could evaluate the incremental effects of added mortality with each new dam. We use the terms "pristine" and "pre-dam" in reference to the survival juvenile and adult salmon (excluding harvest) would have experienced during their migration through the mainstem Snake and Columbia rivers prior to the development of mainstem dams and hatcheries. We found slight differences between the studies cited in their measures used for Parents and Recruits, so we adjusted them, to pre-dam conditions in which,

Parents = successful spawners

**Recruits** = adults that would be caught in the ocean or return to the mouth of the Columbia River.

A discussion of our adjustments follows.

We made adjustments to account for four factors: (1) smolt loss per dam, (2) harvest, (3) adult loss per dam, and (4) prespawning survival (Table 5-1). Parent and Recruit data for the Columbia River used by Chapman et al. (1962) were fully adjusted for harvest in the ocean and river, but were not adjusted for losses of juveniles or adults passing Bonneville Dam, and they were not adjusted for prespawning mortality. Parent and Recruit data for the Deschutes River used by Lindsay et al. (1989) were adjusted for prespawning mortality, but were not adjusted for harvest or for losses of juveniles and adults passing two dams. Lindsay et al. (1989, Table F-1, p. 78) presented recovery data for Coded Wire Tagged (CWT) groups of wild spring chinook smolts in the Deschutes River for the 1977-79 broods, and that data showed a 10% harvest rate in the ocean and a 15% harvest rate in the mainstem Columbia River. Therefore, we applied these harvest rates to their data on Recruits. Parent and Recruit data for the John Day River used by Lindsay et al. (1985) were adjusted for harvest in river and for prespawning mortality, but not for ocean harvest or losses of juveniles and adults passing two dams. Data from CWT recoveries of John Day spring chinook were scant (insufficient marked fish) so we assumed ocean harvest rate was 10%, the same as for Deschutes spring chinook. The net result of these adjustments was that values of  $\alpha$  ranged from 10.5 for the Columbia River to 16.9 for the Deschutes River (Table 5-1). We concluded from this comparison that an  $\alpha$  value of 12 was reasonable for spring chinook in the Snake River. This would be equivalent to an  $\alpha$  value of 7.8 for the Parent and Recruit measures that Chapman et al. (1982) used.

Table 5-1. Reported values of the Ricker  $\alpha$  value for Columbia River spring chinook, and adjustment factors that we applied to make their units of measure equivalent. Adjustment factors are actually estimated survival rates. Values of 1.0 indicate that the specified survival factor was already accounted for.

Basin	Author	Brood Years	alpha	Smolt Dam Loss	Ocean Harvest	River Harvest	Adult Dam Loss	Prespawn Survival	Adjusted Alpha
Columbia	Chapman et al. 1962	38-58	6.8	0.85	1.00	1.00	0.95	0.80	10.5
Deschutes	Lindsay et al. 1989	75-81	0.4	0.72	0.90	0.85	0.90	1.00	16.0
John Day	Lindsay et al. 1985	59-69	6.5	0.72	0.90	1.00	0.90	1.00	11.1

#### 5.1.4 Carrying Capacity and the Ricker $\beta$ Parameter

We estimated the  $\beta$  value of the Ricker curve for Snake River spring/summer chinook from the carrying capacity of the basin for smolts, as estimated in the System Planning Process (Table 5-2). Carrying capacity was actually estimated in the System Planning Process for parr in the fall rather than smolts in the spring, so we applied a 30% overwinter survival rate for parr (see Section 5.3.1) to estimate the carrying capacity for smolts. We assumed that production of spring chinook was limited by rearing habitat, such that, the maximum size of the population is limited by the capacity of the river to produce smolts. In order to make use of the smolt carrying capacity value for estimating  $\beta$ , we defined smolts to be the Recruits. Thus, our estimate of smolt capacity became the value of  $R_m$  in equation (4), introduced previously.

Table 5-2. Estimated spring chinook parr capacities for subbasins in the Snake River drainage. (Data from Integrated System Plan, June 1991)

Race and Location	Parr Capacity
Idaho Spring Chinook	
Accessible Areas	14,467,620
Above Hatchery Weirs	2,490,896
Idaho Summer Chinook	
Accessible Areas	5,479,466
Above Hatchery Weirs	497,666
Oregon & Washington = 10.3% of Idaho	2362,364
<b>SNAKE RIVER BASIN TOTAL</b>	<b>25,297,934</b>

### **5.1.5 Characteristics of the Stock Recruitment Function for Snake River Spring Chinook**

The stock-recruitment function for the Snake River spring chinook, based on its Ricker parameter values and the survival rates specified previously, indicates that the pre-dam population would have had a maximum recruitment of 500,000 chinook produced by about 200,000 spawners (Figure 5-1). Maximum sustainable yield would have been about 330,000 fish produced by about 150,000 spawners (Figure 5-1). As sources of mortality increased, such as from passage mortality at dams, maximum recruitment and maximum sustainable yield would have declined (Figure 5-2). Harvest and mortality have the same effect on surplus yield.

A noteworthy characteristic of the stock-recruitment function is the precipitous drop in surplus production as harvest rate increases beyond that which produces MSY (Figure 5-3). Similarly, the percentage of the habitat's carrying capacity that will be used declines sharply as harvest rate increases beyond that which produces MSY (Figure 5-3). Passage mortality has the same effect as harvest. Once the harvest rate, or the mortality rate over and above the pristine level, mounts up to more than about 80%, the sustainable yield (or surplus production) drops steeply. Population collapse is reached when the harvest and added mortality rates reach slightly above 85% (Figure 5-3). This is highly relevant to de-listing criteria, because it indicates that only a few percentage points of change in mortality can mean the difference between a population in collapse and a population that is near carrying capacity. Thus, a substantial increase in abundance of spawners should be achieved before we can be confident that the population is safe from extinction. We will discuss this further under the section 8.1.2 simulation of stock rebuilding.

### **5.1.6 Stock-recruitment Functions for Population Subunits Based on Habitat Quality**

As discussed under section 2.2, the dynamics of the spring/summer chinook population is likely to differ between geographic subunits in which habitat quality differs. Differences in habitat quality that affect survival rates at any life stage will also affect the  $a$  parameter (recruits per spawner) of the Ricker function. Population subunits with higher survival rates will produce more recruits per spawner than subunits that produce lower survival rates. When these subunits are mixed together as smolts and adults, they face the same mortality factors along their migration route, but the subunits with lower habitat quality are less able to sustain the mortality, and may decline when the subunits from higher

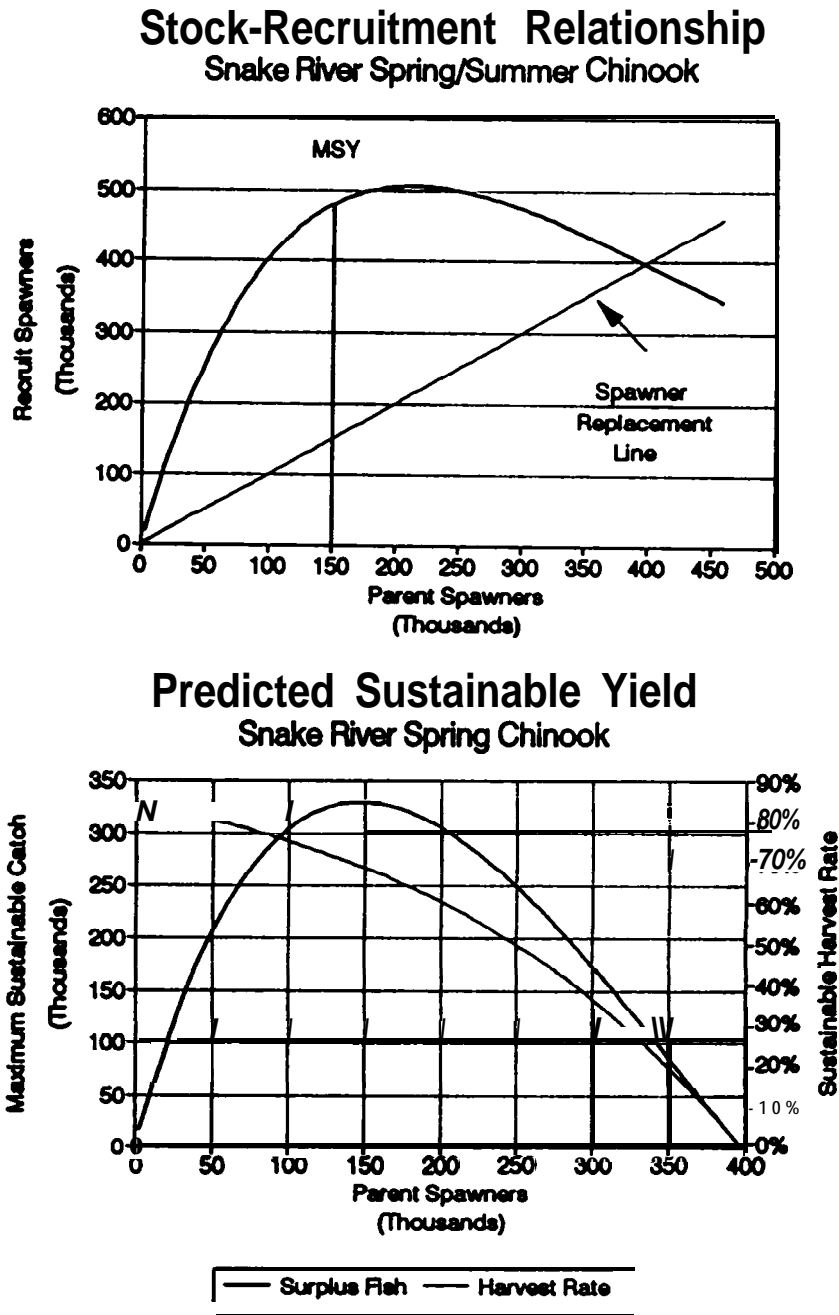


Figure 5-1. Stock-recruitment and yield functions estimated for spring chinook in the Snake River Basin before the advent of hatcheries and mainstem dams.

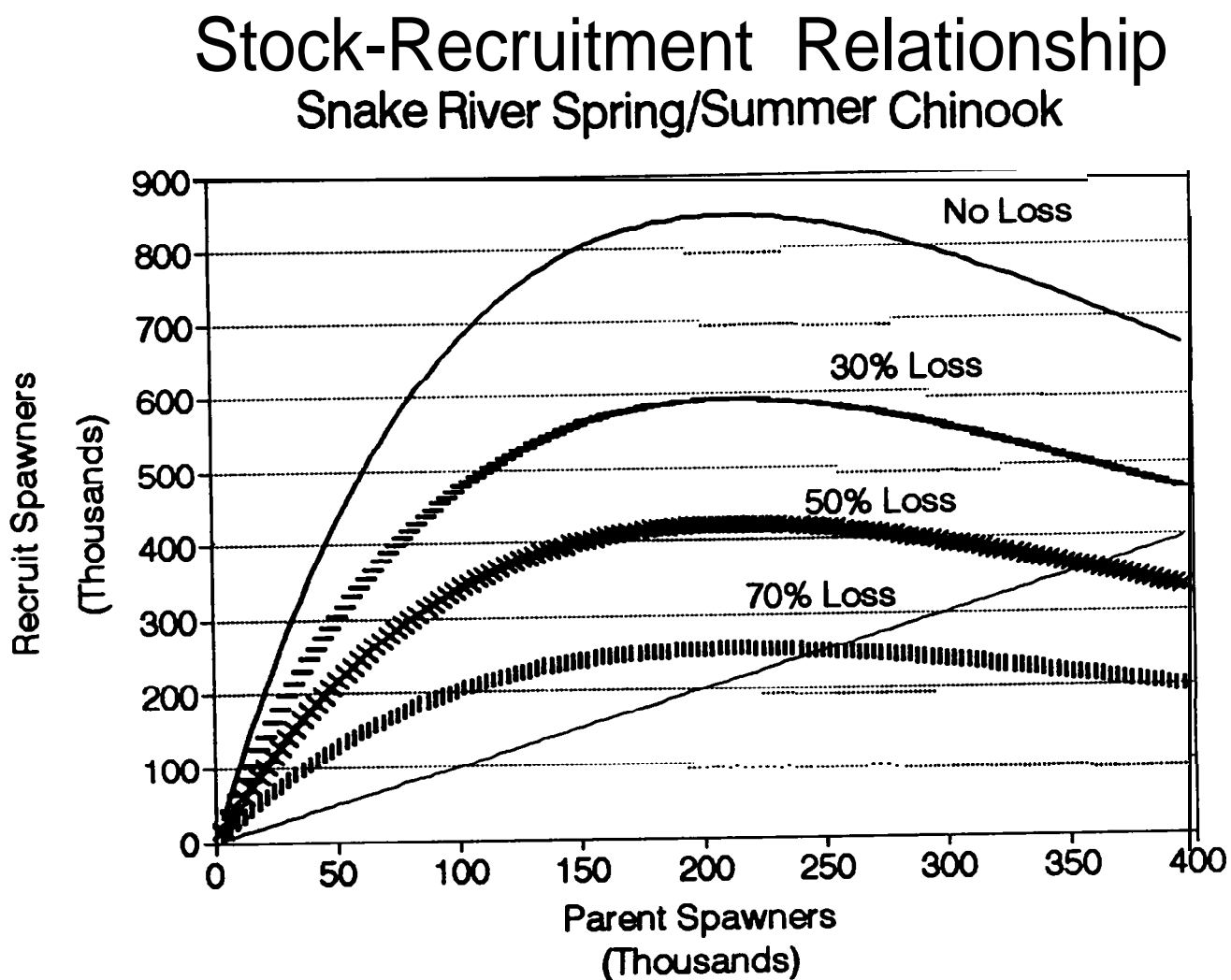


Figure 5-2. Effects of three different levels of mortality on the stock-recruitment and yield functions for spring chinook in the Snake River Basin.



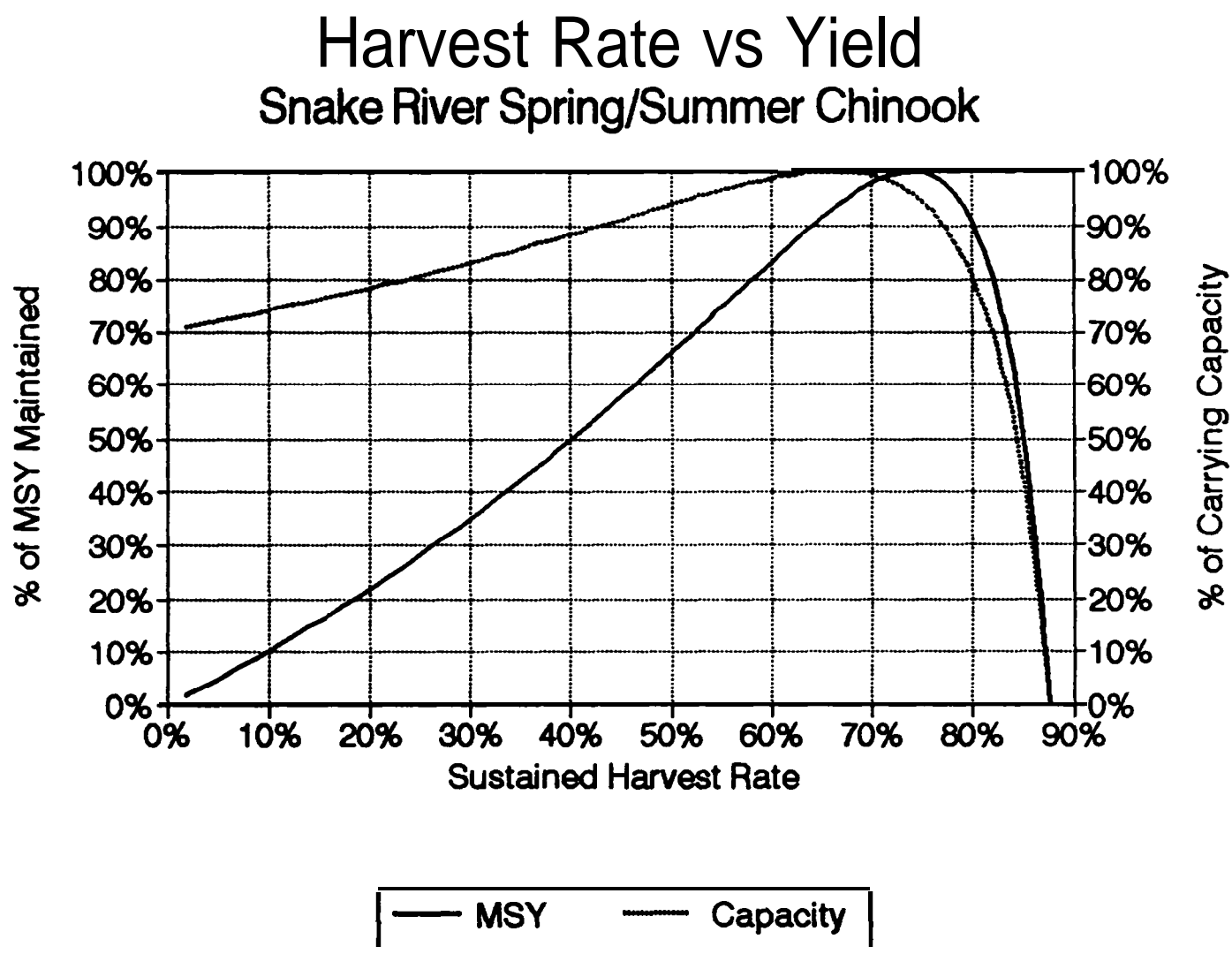


Figure 5-3. Relationship between the proportion of the pristine MSY that can be maintained and the harvest rate plus added mortality. "Pristine" means that passage survival was equivalent to that prior to the advent of hatcheries and mainstem dams.

habitat quality remain stable. This is the same problem as a weak stock in a mixed stock fishery. The magnitude of differences in recruits produced per spawner in poor versus excellent habitat has not been estimated in the Snake River, but we discuss here an example of dividing the population into four subunits, each with its own stock-recruitment function.

In this example, we divided the population into four subunits that correspond to the four habitat quality ratings used by IDFG (1992). The four quality ratings are defined by IDFG as follows (assumed carrying capacity per surface area shown in parenthesis):

Excellent - undisturbed C channels (108 parr/100m<sup>2</sup>)

Good - undisturbed B channels with moderate gradient (77 parr/100m<sup>2</sup>)

Fair - high gradient undisturbed B channels, degraded C channels (44 parr/100m<sup>2</sup>)

Poor - degraded C and B channels (12 parr/100m<sup>2</sup>)

C channels are defined as laterally unconfined with less than 1.5% gradient, such as occurs in meadows. B channels are defined as laterally confined with 1.5% to 4% gradient. The carrying capacity for excellent habitat was derived from field studies in natural streams (Petrosky and Holubetz 1988). We chose these subunits, largely because IDFG has estimated the parr carrying capacity for each habitat rating. We also believe that habitats of a given quality are more likely to have similar recruitment rates between streams than habitats of different quality within the same stream. For example, as the spring/summer chinook populations have declined in the Snake River Basin, spawner numbers have declined more rapidly in stream sections with poor or fair rated habitat, while spawners continue to return to habitats with good or excellent ratings. As additional evidence of higher survival rates in higher quality habitat, annual monitoring of parr densities in Idaho streams by IDFG has demonstrated that excellent quality habitat is consistently seeded with parr at a higher percentage of the carrying capacity than habitat with lower ratings (Figure 5-4).

We chose Ricker  $\alpha$  and  $\beta$  parameters for each of the four subunits such that when the subunits were combined, they would produce an overall population with similar parameters to those we estimated previously for the population as a whole. Choice of the  $\alpha$  parameters (Table 5-3) was somewhat arbitrary, but was intended to reflect real possibilities. The  $\beta$  parameters were estimated as described in section 5.1.4, but the parr capacity for each habitat quality subunit was set according to the proportion of total rearing area that was rated with that quality, weighted by the rearing densities assumed by IDFG

(1992) for that quality (Table 5-3). As shown, 60% of the habitat was rated as poor or fair, but only 40% of the parr capacity was contained in those habitat ratings.

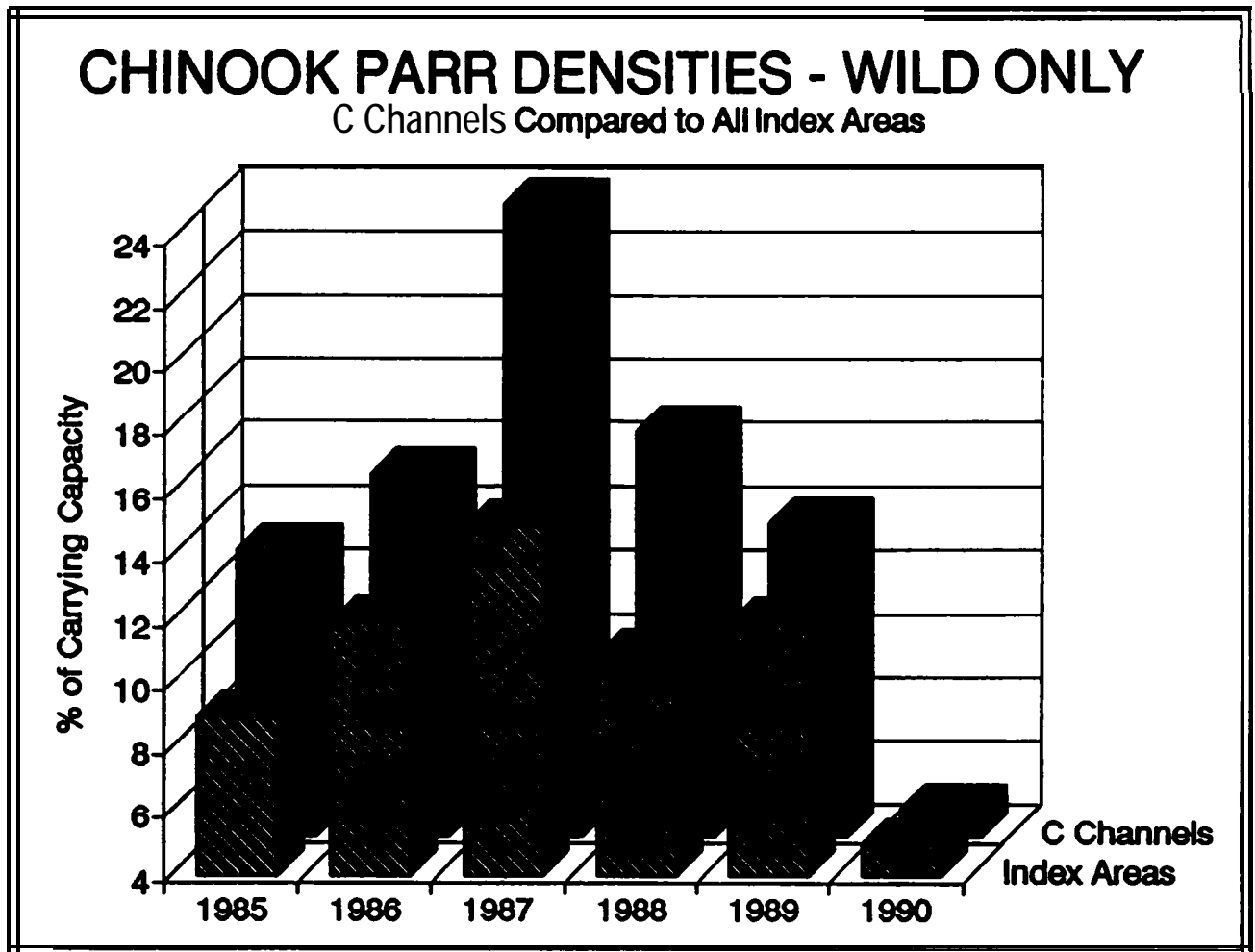


Figure 5-4. Mean percentage of chinook parr carrying capacity that was maintained each year in C channels compared to that in all index areas for wild fish (Rich et al. 1992).

**Table 5-3. Ricker parameters and parr carrying capacities of four population subunits divided according to habitat quality ratings given by IDFG (1992).**

Quality Rating	% of Area	Parr Capacity	$\alpha$ Parameter	$\beta$ Parameter
Excellent	5%	2,500,000	14	$6.67 \times 10^{-5}$
Good	35%	12,600,000	10	$9.45 \times 10^{-6}$
Fair	45%	9,300,000	6	$7.68 \times 10^{-6}$
Poor	15%	800,000	4	$5.95 \times 10^{-5}$

A plot of the stock-recruitment functions for each of these subunits illustrates the substantial differences between subunits (Figure 5-5). A steeper slope on the ascending limb of the curve indicates a greater capacity to withstand harvest or mortality. The harvest rate at which MSY would be achieved is 56.5%, 67.5%, 78.7%, and 84.4% for the poor, fair, good, and excellent habitat subunits, respectively. These differences indicate that MSY for the excellent habitat subunit could not be achieved without substantially over-harvesting the poor and fair habitat subunits.

Before drawing further inferences from this example of four population subunits, we wanted to determine if the combined output of recruits from these four subunits over time would compare with the output of recruits from the single stock-recruitment function we estimated for the pooled population. We tested for this by subjecting both sets of functions (combined and pooled) to a simulation of the estimated harvest rates and dam-passage mortality rates over the 58 year period from 1934 to 1992 (see section 8.1 on Rebuilding for an explanation of the simulation). The results showed that when we used the simulated data on Parents and Recruits to calculate the parameters of the Ricker function, the two curves were similar:

		$R$
Pooled data - one unit	10	$-4.7 \times 10^{-6}$
Sum of 4 Subunits	9.4	$-5.4 \times 10^{-6}$

The similarity of these parameters confirms it is plausible that the Ricker parameters for Snake River spring/summer chinook may have been estimated from the summation of several heterogeneous subunits such as those used in this example. We discuss the implications of this finding further in section 8.1 on Evaluation of Rebuilding Schedules.

## Stock-Recruitment Relationship

### Population Subunits by Habitat Quality

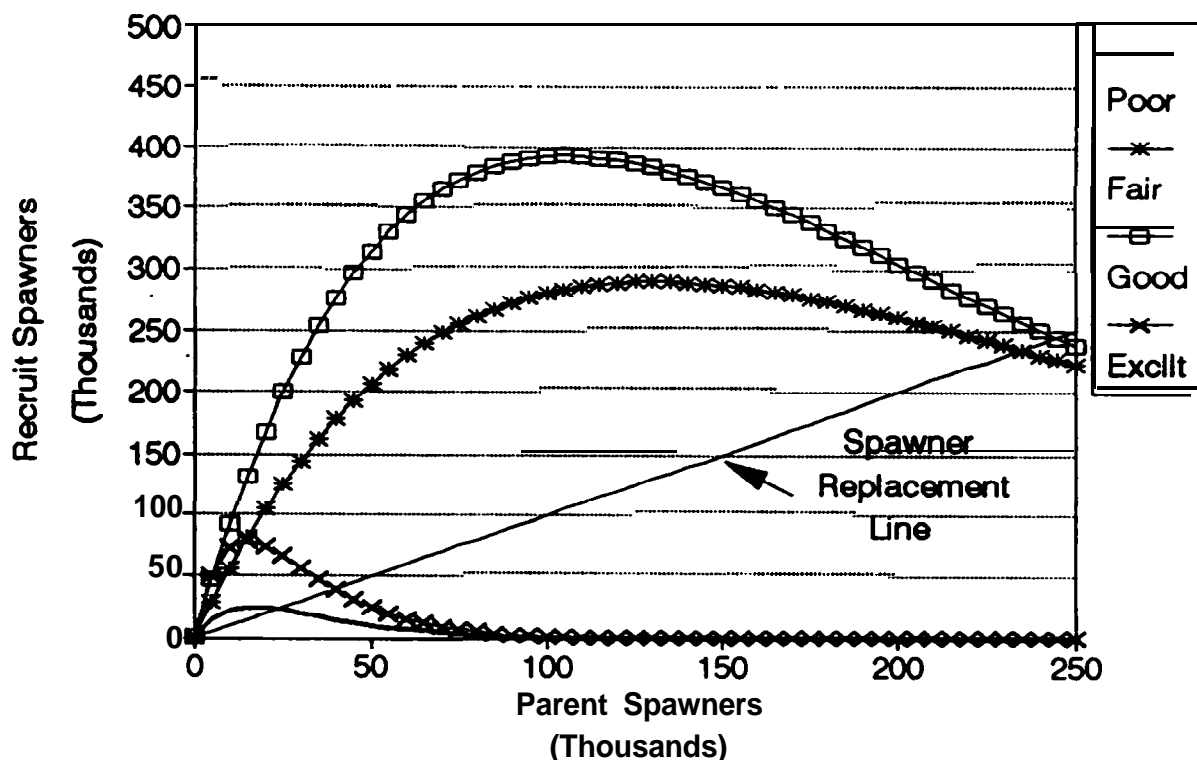


Figure 5-5. Stock-recruitment relationships for four possible subunits of the Snake River spring/summer chinook population. The subunits correspond to habitat quality and their carrying capacity for parr, as estimated by IDFG (1992).

## 5.2 ABUNDANCE MEASURES

### 5.2.1. **Monitoring and Analysis**

In this section we look at monitoring issues and issues related to the assessment of adult counts at Lower Granite Dam (LGR), redd counts, parr density estimates, juvenile passage at LGR, and demographic measures. We recommend that historic LGR wild adult counts not be used as an abundance base. Further, we recommend that future redd counts and parr densities, not dam counts, be the major measures used for evaluating the recovery program. New sampling strategies are recommended that augment the currently sampled index areas with randomly sampled areas. We recommend that population subunits within the Snake River basin be the focus of evaluation and that the subunits be defined, by demographic characteristics and habitat rating, and that standard procedures be followed by all agencies in measuring these demographic characteristics.

#### 5.2.1.1 Lower Granite Dam Adult Counts

##### **Reconstructed Dam Counts**

There are no direct wild fish counts made of spring/summer chinook at Lower Granite Dam because the total hatchery production in the Snake River basin has not been marked. Since hatchery production began in the early 1960's and Lower Granite Dam began in 1975, the adult returns to LGR have always had a hatchery component.

NMFS has, therefore, relied on reconstructed dam counts based on hatchery returns, tribal catch, weir counts and redd counts. The pre-decisional ESA 1977 through 1992 reconstructed wild counts are given in Appendix 3. The current method of reconstruction is outlined below:

- ▶ Allocation of Fish to Wild or Hatchery Origin

Hatchery return allocation to hatchery origin. All hatchery returns are assumed to be of hatchery origin.

Tribal fish catch allocation to hatchery and wild origins: Separate estimates of wild and hatchery catch are reported.

**Weir fish count allocation to wild and hatchery origin:** The number of CWTs (coded wire tags) recovered from the weir are divided by the mark rate to obtain the estimated number of hatchery fish comprising the weir count. The remaining weir count is the estimated wild count.

- **Allocation of Redds to Wild or Hatchery Origin**

**Below-weir redd allocation to wild and hatchery spawners:** The wild and hatchery proportions estimated from weir counts is multiplied by the number of redds counted below the weir to respectively estimate the numbers of hatchery and wild redds.

**Above-weir redd allocation to wild redds:** counts from index areas above weirs are treated as wild/natural.

- ▶ **Expansion of Fish and Redd Counts**

**Prespawning survival adjustment of fish counts:** Hatchery returns, weir counts, and tribal catch estimates are assumed to have an 80% survival rate from Lower Granite Dam; therefore wild and hatchery allocated fish counts are multiplied by  $1.25 = 1/.8$  to express them as LGR equivalents.

**Redd count expansion:** All redd counts are expanded by 5.4 to obtain LGR equivalents.

- ▶ **Application to Dam Count**

**Estimating wild proportion of LGR adult passage:** The wild and hatchery LGR equivalents obtained from the above enumerations and expansions are totalled and the wild LGR equivalent is divided by this total to estimate the wild proportion.

**Estimating LGR wild count:** The actual dam count, excluding jacks, is multiplied by the estimated wild proportion to obtain the estimated LGR wild count.

**NOTE:** The above process is normally applied separately to the spring and summer portions of the run. In 1992 the estimated wild proportion was based on wild and hatchery totals pooled over the spring and summer portions of the run. This pooled proportion was then applied separately to the spring and summer dam counts. From 1977 through 1981 the whole of the summer portion of the run was assumed to be of wild origin, presumably because the McCall hatchery's summer chinook production began in 1980 (Chapman et al., 1991).

As an Abundance Measure. The reconstructed counts are likely to be biased, and possibly very biased, estimates of wild abundance. The following will contribute to the bias:

- ▶ **Bias in allocation of below-weir redds:** The wild allocation of below-weir counts requires the assumption that both hatchery and wild fish will be equally successful in digging redds before the time of the redd count. If the hatchery success differs from that of wild, then the assignment of redds to wild and hatchery sources will be biased. This bias applies to a small portion of the total basin redd count because most redd index areas were above the weirs; of the 1537 redds used to determine the wild proportion in 1992, only 212 (14%) were below-weir counts (Reconstructed count spreadsheet for 1992 provided by Debbie Watkins, BPA, Portland).
- ▶ **Bias in allocation of above-weir counts:** The assignment of all above-weir hatchery redds to the wild/natural category requires the assumption that all spawning fish released above or escaping the weir are wild/natural fish. All Clearwater and Salmon subbasin spring chinook are enumerated as wild LGR equivalents under this procedure, even though the Idaho Department of Fish and Game (IDFG) separates index areas for both spring and summer chinook into wild and hatchery-influenced areas (Appendix 6). The failure to allocate a portion of these hatchery-influenced redds to hatchery production will contribute to an under-estimation of the LGR wild count. This bias applies to a large portion of the total redd count.
- ▶ **Bias in the 5.4 expansion factor for redds:** We believe the 5.4 expansion may be far too small. Since 1986, ODFW has been making additional redd counts within index streams but outside the historic index areas within the Imnaha and Grande Ronde subbasins. These outside-index-area counts have been made in conjunction with the standard within-index-area counts (Jonasson, et al. 1992). When averaged over years, the pooled within- and outside-index-area count for the Imnaha subbasin was



greater than the within-index-area count by a factor of 1.52; for the Grande Ronde, the factor was 1.49 (Table 5-4). Further, some of ODFW's historic index areas have been surveyed at intervals following the standard index-survey time to determine whether additional redds had been dug (Jonasson, et al. 1992). The yearly averages of new counts accumulated over the survey times were 1.90 and 1.79 times the standard index-survey-time counts for the respective subbasins (Table 5-5). Multiplying the new area and the standard index-time adjustment factors together gives 2.88 for the Imnaha and 2.67 for the Grande Ronde subbasins. If one assumes 2.0 adults per redd and a 0.8 survival from Lower Granite to the spawning grounds, the expansion factor to produce LGR equivalents become 7.2 and 6.7 for the respective subbasins. The assessment was applied to the individual years for each of the two subbasins. More than seventy-five per cent of the estimates exceeded 5.4. Even with the additional enumerated areas, there is no redd enumeration in much of the subbasins' potential spawning habitat; therefore the 7.2 and 6.7 expansions are still too small. If the degree of under-count in the Salmon and Clearwater subbasins is similar to that in the Imnaha and Grande Ronde subbasins, then the presently used 5.4 expansion greatly underestimates the LGR wild counts.

7 Relative proportional allocations to spring and summer proportions of the run will be biased. It appears that all Imnaha fish and redd counts are allocated to the spring portion of the run. Both the summer and spring portions of the run contribute to the Imnaha and Grand Ronde (Bjornn et al. 1991). Further, the 100% allocation of the summer portion of the run to wild fish prior to 1982 is biased (refer to Appendix 3). Pahsimeroi Hatchery began operation in 1967 and has utilized the summer run for part of its broodstock, and releases into the Pahsimeroi River and into the South Fork of the Salmon River from 1974 are identified as summer chinook (Appendix 3 of Chapman et al. 1991). If all hatchery returns to Pahsimeroi Hatchery are allocated to the spring portion of the run, then this allocation will contribute to an under-estimation of the wild spring proportion and an over-estimation of the wild summer proportion.

Table 5-4. Number of redds inside and outside index areas of the Imnaha and Grande Ronde subbasins 1986-1992.

		1986		1987		1988		1989		1990		1991		1992	
Index Area		Miles	Redd Count	Miles	Redd Count	Miles	Redd Count	Miles	Redd Count	Miles	Redd Count	Miles	Redd Count	Miles	Redd Count
Imnaha Subbasin															
Big Sheep Creek	Inside	4.0	15	4.0	3	4.0	14	4.0	1	4.0	0	4.0	1	4.0	0
	Outside	28.5	10	9.0	13	9.0	0	5.0	1	5.0	2	5.0	5	5.0	3
Imnaha River	Inside	9.7	127	9.7	112	9.7	135	9.7	40	9.7	33	9.7	51	9.7	75
	Outside	22.5	37	22.5	28	22.5	37	25.9	32	25.9	11	25.9	41	25.5	42
TOTAL	Inside	13.7	142	13.7	115	13.7	149	13.7	41	13.7	33	13.7	52	13.7	75
	Outside	49.0	47	31.5	41	31.5	37	30.9	33	30.9	13	30.9	46	30.5	45
(INSIDE + OUTSIDE)/INSIDE			1.33		1.36		1.25		1.80		1.39		1.88		1.80
1986-92 Mean										(INSIDE + OUTSIDE)/INSIDE					1.52
Grande Ronde Subbasin															
Lostine River	Inside	3.0	48	3.0	49	3.0	107	3.0	20	3.0	16	3.0	11	3.0	14
	Outside	22.0	13	9.0	19	14.0	57	14.0	5	14.0	3	14.0	9	14.0	8
Grande Ronde River	Inside	8.5	37	8.5	112	8.5	99			8.5	4	8.5	10	8.5	97
	Outside	13.7	11	13.7	42	13.7	5			13.7	0	13.7	0	13.7	1
Sheep Creek	Inside	6.0	4	6.0	7	6.0	0	6.0	0	6.0	0	6.0	0	6.0	5
	Outside	3.0	2	3.0	2	3.0	15	3.0	1	3.0	1	3.0	0	3.0	2
Catherine Creek	Inside	7.5	47	7.5	103	7.5	99	7.5	31	7.5	19	7.5	15	7.5	36
	Outside	7.0	11	7.0	46	7.0	33	7.0	4	7.0	2	7.0	0	7.0	1
South Fork Catherine Creek	Inside	2.0	21												
	Outside	4.0	0												
Wenaha River	Inside			6.0	62	6.0	98	6.0	9	6.0	31	6.0	28	6.0	58
	Outside			22.0	86	22.0	85	15.5	9	15.5	47	16.5	27	15.5	110
Minam River	Inside													13.5	16
	Outside													5.0	0
TOTAL	Inside	27.0	157	31.0	333	31.0	403	22.5	80	31.0	70	31.0	84	44.5	226
	Outside	49.7	37	54.7	195	59.7	175	39.5	19	53.2	53	53.2	36	58.2	122
(INSIDE + OUTSIDE)/INSIDE			1.34		1.59		1.43		1.32		1.76		1.56		1.54
1986-92 Mean										(INSIDE + OUTSIDE)/INSIDE					1.49

Table 5-5. Number of redds at standard and subsequent enumeration times. Imnaha and Grande Ronde subbasins 1986-1992.

		1986		1987		1988		1989		1990		1991		1992	
Index Area	Miles	Standard Time	Over All Times	Standard Time	Over All Times	Standard Time	Over All Times	Standard Time	Over All Times	Standard Time	Over All Times	Standard Time	Over All Times	Standard Time	Over All Times
Imnaha River															
segment 1	7.7			27	86	100	122	24	27	29	59	39	56	56	83
segment 2	3.5													16	76
TOTAL				27	86	100	122	24	27	29	59	39	56	76	159
(All/Standard)					3.52		1.22		1.13		2.03		1.41		2.09
1986-1992 Mean (All/Standard)														1.90	
Grande Ronde Subbasin															
Humaine Creek	1.3	5	11	5	25	5	30	2	19	0	24	4	12	0	14
Lostine River	3.0	46	77	27	53	107	125	20	47	16	24	11	19	14	26
Grande Ronde River															
segment 1	5.0	19	33	14	65	22	33					10	10	21	29
segment 2	3.0									3	21			76	89
Catherine Creek	2.0	25	32	15	45	37	40	6	12	7	9	9	10	16	25
Minam River	4.0			61	75	41	50	19	24	36	50	13	26	1	60
Wallowa River	4.0													0	4
Wenaha River	6*											14	15	16	15
South Fork Wenaha River	6.0													55	65
Prairie Creek	1.0													1	3
* 6 miles 1991, 5 mile subsegment 1992															
TOTAL		97	153	122	283	212	296	49	102	62	126	61	66	205	352
All/Standard			1.58		2.16		1.41		2.05		2.06		1.39		1.72
1986-1992 Mean (All/Standard)														1.79	

Source: Rhine Messmer (ODPW) 1993

There are probably other biases.

The historic and current reconstructed LGR wild count should be regarded as a biased estimate of wild abundance. If the current pre-recovery LGR wild counts are underestimates, and if these counts are to serve as the pre-recovery base of comparison, then the future recovery program might erroneously be judged as a success because future estimates would be compared to historic under-estimates giving a false measure of increase.

**RECOMMENDATION: Unless the above sources of bias can be measured and adjusted for, wild dam count should be regarded as a poor measure of historic and current abundance and should not form a reference base for assessing the recovery program.**

As a Trend Measure. Trend measures based on historic and current LGR wild counts are likely to be **biased**, and possibly very biased estimates of true trend. Unbiased estimates of trend in wild abundance are possible in the presence of biased abundance estimates under only two conditions:

- the actual proportion of wild remains constant over time, or
- the relative biases for hatchery and wild are equal

The conditions required for equal bias are not likely to hold for Snake River spring-summer chinook.

- **The actual proportion of wild remains constant over time:** The proportion of wild comprising the dam count will likely change over time. During the historical eight-dam era, the estimated hatchery component has been increasing. During the recovery era, that proportion may continue increasing if a supplementation program is to be a major component to the recovery effort. If the recovery program is successful, then the hatchery component would eventually begin to decrease and perhaps to level off.
- **The relative biases for hatchery and wild are equal:** The bias associated with the hatchery is primarily the bias on the expansion of fish counts by 1.25. The bias associated with the wild is primarily the bias in the redd count expansion by 5.4.

These biases are unlikely to be the same. The biases that we identified in the previous section probably impact the wild more than the hatchery reconstructed counts.

The nature of the bias is demonstrated in Figure 5-6 for a hypothetical population in decline with a decreasing proportion made up of wild fish. Declines in both the total dam count and in the wild proportion of the dam count characterize the eight-dam era. The formula used for computing wild proportions from reconstructed hatchery and wild counts with different biases is presented in Appendix 7. The bias effects illustrated in the figure apply to a population experiencing a yearly 2% reduction in total return and a 5% yearly reduction in the wild proportion of the run. These reductions result in a given year's true wild count being 93% of the previous year's [ $0.93 = (1-0.02)(1-0.05)$ ], or a yearly reduction of 6.9%. In the figure, for the case of the wild bias being 4 times that of hatchery, the average yearly reduction is approximately 3.9%, much lower than the actual 6.9% reduction. For the case of the wild bias being one-fourth that of hatchery, the average yearly reduction is 10.5%, much higher than the actual.

**RECOMMENDATION:** Unless the sources of bias identified above can be measured and adjusted for, trend estimates based on reconstructed counts should be regarded as being poor estimates.

**Estimation of extinction probabilities.** Extinction models, whether they are analytical or simulation models, would probably involve wild counts at time  $t$  as a function of wild counts at time  $t-\Delta t$ . Some parameters in the models are likely to be estimated using historical data. Biases associated with LGR wild counts at times  $t$  and  $t-\Delta t$  will probably differ substantially if the wild proportion is changing over time; therefore, reconstructed dam counts should not be regarded as a suitable base for estimating extinction model parameters.

**RECOMMENDATION:** Unless the sources of bias can be measured and adjusted for, extinction model parameter estimates based on reconstructed counts should be regarded as being poor estimates.

**General Evaluation.** The reconstructed wild LGR counts depend almost solely on the expansion of redd counts. Current estimates of tribal catch contribute only minimally to the reconstructed wild count. Therefore, rather than consider the biased reconstructed dam counts as measures to estimate abundance, trend, or extinction probability, redd counts should be the main focus of any historical evaluation.

### BIASED WILD COUNT WHEN RELATIVE BIAS OF WILD TO HATCHERY IS 0.25, 1 AND 4

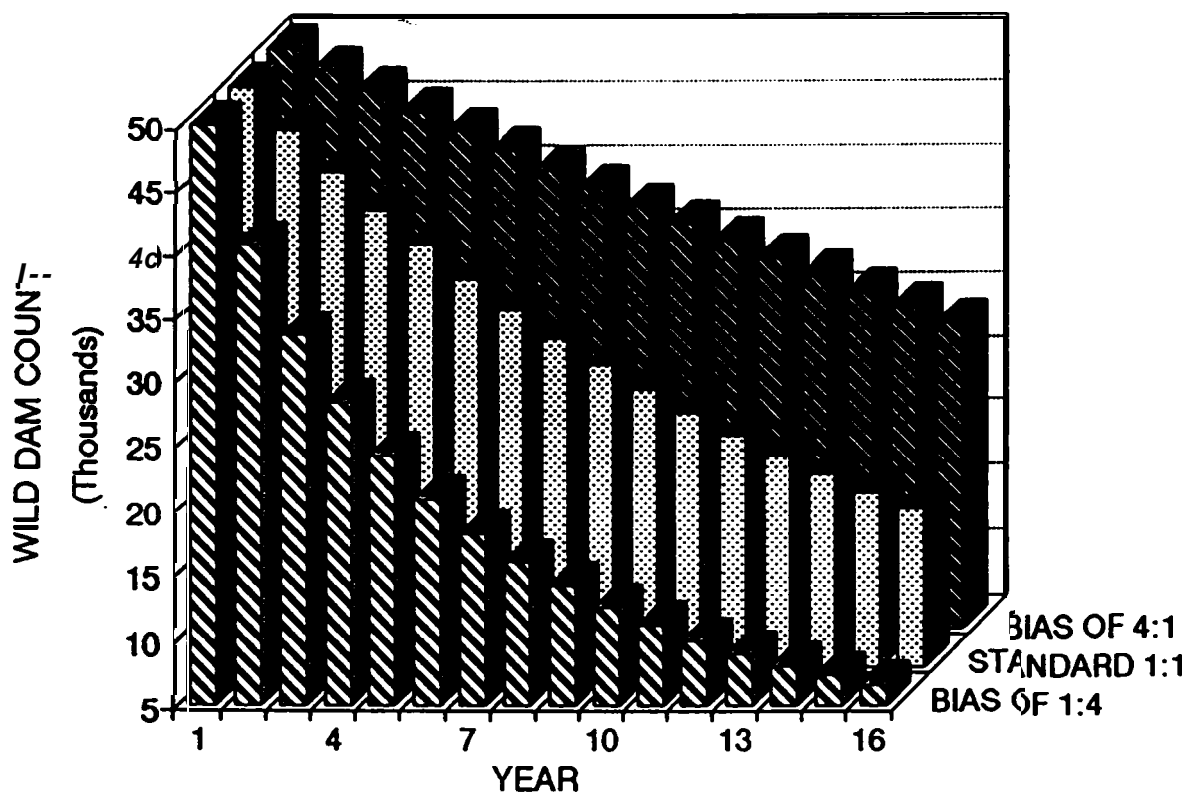


Figure 5-6. Biased wild count when relative bias of wild to hatchery is 1:4, 1:1 (unbiased standard), and 4:1.

**Future Dam Counts**

Beginning in 1993, all total hatchery releases are to be marked. Therefore, it should be possible to obtain adequate measures of abundance, trend, and extinction applicable to those counts.

These measures based on future counts can only assess the population of wild/natural returns without identifying whether or not their parents were wild/natural spawners. If the recovery program is to be successful, then the trend of interest should be whether the natural recruits derived from natural/wild parents are increasing. If not, then any increase in natural production will always be dependent on the hatchery program; we should not consider a population for de-listing unless there is sufficient evidence that the natural population could be self-sustaining. Dam counts cannot be used to provide this information.

This does not preclude dam counts being used as a measure of success for some components of the recovery program, and we discuss the estimation of dam counts under different marking and enumeration scenarios.

**Marks Visible through Viewing Window.** Probably the only easily and economically applicable mark that can be consistently read through the viewing window is the clipped adipose fin. We emphasize the need for de-sequestering this mark's use from the coded wire tag.

If the adipose fin clip can be used, then the counts would be estimates of wild passage under current enumeration conditions. The counts are not a census; that is, not all fish are being counted. Counts are made during day-light shifts, 50 minutes within each 60 minute period (D. Rawding, WDW, personal communication). These counts are expanded by  $1.2 = 60/50$  to obtain the day-light hour count. There is no expansion of the day-light count to obtain 24-hour estimates of the abundance. The day-time shifts vary over the year. The shift schedule for LGR in 1992 is given in Table 5-6.

**Table 5-6: Counting shifts for adult passage at Lower Granite Dam in 1991 (Source: Corps of Engineers 1992.)**

Period	Shift (Pacific Standard Time)	Ladders in service?
1 Jan - 2 Jan	No counting	Yes
3 Jan - 23 Jan	No counting	No
24 Jan - 28 Feb	No counting	Yes
1 Mar - 31 Mar	8:00 A.M. - 4:00 P.M.	Yes
1 Apr - 31 Oct	4:00 A.M. - 8:00 P.M.	Yes
1 Nov - 15 Dec	6:00 A.M. - 4:00 P.M.	Yes
16 Dec - 31 Dec	No counting	Yes

In 1992 the Columbia River Inter-tribal Fish Commission conducted a field test to evaluate feasibility of using time-lapse video technology. Twenty-four hour counts were made from 1 June through 15 December. The images were high quality, and cost comparisons indicated that video counts were less than half the cost of the current counting method (Douglas Hatch, Columbia River Inter-tribal Fish Commission, Portland, personal communication).

**RECOMMENDATION:** We recommend using the 24-hour video counts in place of viewing-window counts if each fish is clearly visible in its entirety on at least one frame to permit length and mark assessment.

**Marks not Visible through Viewing Window.** If the marks cannot be seen at the viewing window, then it will be necessary to sample the run in order to estimate the proportion of hatchery marks. It can be very difficult to engineer adult sampling facilities which trap and hold a truly representative portion of the population. However, it may be necessary to implement such sampling if contributions of the specific hatchery and supplementation programs requiring the use of unique marks are to be monitored at Lower Granite Dam.



**RECOMMENDATION:** The use of the clipped adipose fin as a mark should be desequestered from the coded wire tag so that the mark can be used to differentiate hatchery fish from the wild/natural listed species.

#### **5.2.1.2 Redd counts**

Redd counts have the potential of being the best indicator of effective spawner number. Multiplying redd counts by a factor that reflects the spawner sex ratio has been used to estimate spawner abundance. In this section we discuss redd counts and in a later section on demographics we discuss the expansion of redds to estimate spawner number.

#### **Historic Redd Counts**

Redd counts from some index areas have been made yearly since before the construction of Ice Harbor Dam. IDFG and ODFW initiated redd count surveys of fixed index areas in the late 1950s/early 1960s. Edited summaries of spring-summer index counts for the Salmon and Clearwater subbasins have been produced (Peter Hassemer, IDFG, Boise, personal communication) and are presented in Appendix 5: Imnaha and Grande Ronde summaries are presented in Appendix 6.

**As Absolute Abundance Measure.** Historic and current index counts are not appropriate for absolute abundance estimates at the subbasin or basin levels. The index areas were originally chosen to monitor change over time; they were not chosen to be representative of the total spawning habitat. Since 1986, ODFW has performed redd counts both inside and outside index areas within selected streams. Pooled index-area counts have been greater than counts from outside of the index areas for both subbasins in each year of evaluation (Table 5-4). This indicates that redd count assessments from index areas would be biased if they were expanded to the subbasin level. Similar biases are believed to exist for IDFG counts as well (Pete Hassemer, IDFG, Boise, personal communication).

The magnitude of the bias is not known. Even when the evaluated areas inside and outside the index areas are taken together, most of the current and potential spawning habitat is not being evaluated. No sampling frame exists for any of the subbasins from which samples can be randomly drawn. Therefore, the sub-basin or basin-wide redd abundance should not be estimated from currently enumerated areas. IDFG and ODFW never intended the index counts to be used for estimating absolute fish abundance; and the Recovery Plan should not rely on the redd count for such purposes. Direct use of index

redd counts will under-estimate the number of redds in the total spawning habitat. Expanding the redd counts by the inverse of the proportion of the total spawning habitat that is enumerated will likely greatly over-estimate the number of redds, and any other expansion will likely produce biased estimates.

The standard time of enumeration does not accurately estimate the number of redds within the index areas. ODFW (Table 5-5) has substantially increased the total number of redds observed by sampling an additional two to three times following the standard time.

**RECOMMENDATION: Historical and current redd counts should not be used as a measure of absolute abundance.**

**As a Trend measure.** Index area counts could be used to assess growth trends. The relative biases associated with index counts may be reasonably constant over time. If this is the case, then the historical and current counts may be used to evaluate trends.

***Summarizing Information over Index Areas.*** The manner in which trend measures are summarized over index areas can lead to different conclusions. This is illustrated in Table 5-7 for Poisson regression fits of the exponential model made on 24 index areas having complete records for the eight-dam era, 1978-1992 (see the table in Appendix 4 for details). Even though nearly 90% of the index areas have negative trends, the decline based on pooled redds is not significant. The mean of the individual trend estimates is highly significant and is much greater in magnitude than the trend estimate based on the pooled count.

The single index area showing significant growth is South Fork Salmon River, and it makes up a majority of the redd count for the 24 index areas evaluated; for example, in 1992 South Fork Salmon River's redd count was 685, this represented 55% of the 1992 redds from the 24 index areas analyzed. Excluding the South Fork Salmon index area resulted in the mean of the estimates and the estimate based on the total count being more consistent. However, the significance level ( $P < 0.01$ ) associated with the mean of trends was still higher than that ( $P = 0.06$ ) associated with the trend of total count. The test on means seems to be more powerful no matter what exponential measure of trend is used (refer to table in Appendix 4).

Table 5-7 Indications of redd decline or growth from 1978-1992 based on 24 index areas with complete count information based on Poisson regression of count on time for exponential model, $\exp(b \cdot \text{year})$		
Trend Measure		Proportion
Proportion of index areas with estimated <b>decline</b> ( $b < 0$ )		21/24 = 0.88
Proportion index areas with <b>significant</b> ( $P < 0.10$ ) <b>decline</b>		10/24 = 0.42
Proportion of index areas with estimated <b>growth</b> ( $b > 1$ )		3/24 = 0.12
Proportion of index areas with <b>significant</b> ( $P < 0.10$ ) <b>growth</b> (South Fork Salmon River showing significant growth)		1/24 = 0.04
Trend Measure	b (yearly % decline)	P
Average trend (average of 24 index area trends)	-0.102 (9.7%)	<0.01
Single trend measure based on pooled redd count over 24 index areas	-0.047 (4.6%)	0.28
Average trend (average of 23 index area trends, excluding South Fork Salmon)	-0.111 (10.5%)	<0.01
Single trend measure based on pooled redd count over 23 index areas excluding South Fork Salmon	-0.093 (8.9%)	0.06

In the absence of random sampling, it is not possible to recommend a generally acceptable method of summarizing the trend. In Table 5-7 the averages presented were simple arithmetic means, each trend being regarded as equally important in characterizing

the whole basin's trend. Weighted means could be used instead. The weights could be total area of the index area, the index area's stream length, or the average number of redds over time. (Note: Using the number of redds as a weight would tend to give estimates similar to the trend of the pooled counts.)

The trend assessments in Table 5-7 were based on summaries over the whole basin. Basin-wide summaries will not be appropriate. The decision as to the level of summarization should be based on factors that are thought to be demographically important, such as whether the spawning habitats are similar in quality, whether the spawners are from the spring or summer portion of the run, whether spawners have similar age and sex distributions, and whether or not the redds are hatchery-influenced.

In the example, the decision to omit the South Fork of the Salmon River was driven by results; such decisions should have been based instead on characteristics that distinguish the South Fork population or habitat from other index areas. This river is the site of a hatchery weir, and there is a long history of summer chinook hatchery releases into the system (Appendix 3 of Chapman et al. 1991). This river suffered severe habitat degradation in the 1960s (Platts and Megahan, 1975 referenced in Petrosky and Schaller 1992). Subsequent habitat improvement together with hatchery stocking may have led to the growth observed in the eight-dam era. The population dynamics associated with this river would not represent those associated with wild or natural production.

Other positive trends observed may be due to increases in population sizes due to hatchery straying. A further evaluation of two of the redd index areas is presented later in Section 5.2.2. that eliminates years for which hatchery straying was known to have been a problem.

**RECOMMENDATION: Based on our investigations to date, trends will be more powerfully assessed using means of trends over index areas rather than fitting a trend to the pooled count. Therefore, we recommend that historical trends be separately estimated for each index area and appropriately averaged for the purpose of statistical tests. This does not preclude trend analysis on pooled counts. The level of trend summarization, whether based on means or pooled totals, should be based on such demographic characteristics which are discussed in a later section.**

*Missing information.* Occasionally, an index area is either not counted or its count is not available. Summary counts presented in reports often appear to be totals that make no

adjustments for such missing data; i.e., a missing data point seems to be treated as a 0 count rather than missing information (e.g., Salmon and Clearwater subbasins, Appendix 5). In some cases the same value seems to be substituted for a series of missing values (e.g., Imnaha Subbasin in Appendix 6) in which case a trend is not being included. If pooled counts are to be used in assessing historical trends they must be adjusted for missing information.

**RECOMMENDATION: Biologically and statistically sound adjustments should be made for missing data if trends are to be based on pooled counts.**

***Estimating Extinction Probability.*** Estimation of extinction model parameters using redd counts may be possible for reasonable groupings of redd index areas.

We mentioned in Chapter 4 that extinction probability models should include variability in the estimated or simulated parameters. As a model validity check, the model's variability in the estimated or simulated parameters may be compared to the variability among the parameter estimates from the different comparable index areas within the same subpopulation.

**RECOMMENDATION: Estimate trend parameters from comparable index areas, and use them to characterize the variance in parameter estimates.**

### **Future Redd Counts**

To assess the impact of the recovery program on redd count abundance and trend,

- new sampling strategies should be adopted,
- assessment should be focused on redds produced by natural spawners, and
- adjustments should be made for environmental variables.

**Stratified Random Sampling.** New spawning ground sampling strategies will have to be developed if spawner abundance is to be estimated on a subbasin or basin basis. We recommend the following:

**Index areas:** The historical index areas should continue to be monitored at the standard time of sampling. Data from these index areas would be used primarily for time trend analyses with the specific goal of comparing recovery period measures to those of pre-recovery period.

**Stratified random sample areas.** Areas should be randomly sampled within subbasin strata covering the whole spawning habitat (e.g., stratified random sampling procedures now being applied to coastal Coho by ODFW; see Jacobs and Cooney, 1991). Such strata should be selected so as to reflect the demographics of the stock and the nature of the habitat.

Within each stratum a subsample of the randomly sampled areas should be systematically re-enumerated over the spawning season to assure that a near peak value of redds has been assessed as has been done by ODFW in certain index areas. It would be ideal if such re-enumeration was possible within each sampled area; however, such an effort may not be cost-effective. To make effective use of survey teams, the times of enumeration could differ over the single-enumerated areas. Such a strategy is illustrated in Table 5-8 for one stratum.

**Table 5-8. Times for redd enumeration in future sampled areas**  
(for a given stratum, X indicates that a given sampled area is evaluated during the given time of enumeration).

Time of enumeration	Sampled Area											
	1	2	3	4	5	6	7	8	9	10	11	...
1	X	X					X		X	X		...
2	X		X		X			X		X		...
3	X			X		X				X	X	...

Single-enumerated area counts could be calibrated within each stratum based on the accumulated counts from the re-enumerated areas. Stratum-level, subbasin-wide and basin-wide redd counts could then be estimated.

For spawning areas that are believed to be under-utilized, sampling should still take place in order to monitor possible expansion of the spawning habitat over the recovery period. However, the sampling effort could be kept low until there is evidence of growing exploitation of the habitat.

Estimates from the index areas and from the stratified random sample can be combined by treating relevant groupings of index areas as if they constituted completely enumerated strata.

Sampling over years: A complete re-sampling from the sampling frame from year to year is not likely to provide data that could be used for precise time-trend estimates because of year-to-year sampling variation. However, it is advisable to include some new units from one year to another to better accommodate spacial variability. The inclusion of new sample units would be required if the spawning habitat increases over the recovery period; however the inclusion of some new units is recommended whether or not the spawning habitat changes.

A rotation schedule could be developed that rotates some previously sampled units out, replacing them with new sample units. In developing crop forecasting estimates, the Agricultural Research Service maintains 80% of its sampled areas from one year to the next, replacing 20% of its sample units with new random samples each year. Each unit is retained for four consecutive years before being rotated out. Overlapping four-year inclusions over sampled units permits more precise time trend assessments than would be possible from complete re-sampling from year to year.

Such a rotational schedule applied to redd count areas might eventually include the index areas, resulting in the rotation of the index areas out of the survey. However, the index areas should not be included in a rotation schedule until there is evidence that time trends can be adequately assessed using the rotation of the stratified random sample units.

RECOMMENDATION: Stratified random sampling should be used in selecting new redd count areas; current index areas should continue to be assessed for the foreseeable future.

**Allocation to Wild and Hatchery Spawners.** IDFG has classified some of its index areas as being comprised solely of natural/wild production. Since all hatchery releases should be marked beginning in 1993, by 1996 it should be possible to tally all spawned-out carcasses into hatchery and natural origin categories.

**RECOMMENDATION:** The number of redds in an index area should be multiplied by the wild/natural proportion of spawned-out female carcasses to estimate the number of wild-origin redds. This estimate should then be the primary focus of abundance, trend and extinction assessments. Assessments based on total redds within enumerated areas would still continue because the pre-recovery reference base would have been based on total redds per index area.

**Adjustment for Environmental Effects.** As mentioned earlier, the pre-recovery period being considered by NMFS as a base for comparison is 1986-1990. Returns and out-migrants from that period would have been affected by drought. It is important that the effect of drought be adjusted for in order to protect against attributing an increasing wild fish population to the recovery program when the increase was attributable to better climatic conditions.

We earlier suggested including a climatic indicator as a covariate. If climatic conditions improve over the recovery period, then any recovery trend should be less pronounced when adjusted for the climatic indicator. The decline during the historic eight-dam era can serve as an example. Climatic conditions were worse toward the end of the period. The decline could have been partially attributable to the poor water years; if so, then adjusting the decline for an appropriate climatic indicator should result in a smaller decline. Assuming that returns are made up primarily of fish that out-migrated two or three years previously, we used the following values for climatic-indicator covariate,  $x$ :

$x = 1$  if both age 4 and 5 returns outmigrated during drought years,

$x = \frac{1}{2}$  if only one or the other of the age 4 and 5 returns outmigrated during a drought year, or

$x = 0$  if neither of the age 4 or 5 returns outmigrated during a drought year.

The affect of the adjustment on the trends for the eight-dam era redd counts from the 24 redds having complete data sets are summarized in Table 5-9.



**Table 5-9 Comparisons of trend measures unadjusted and adjusted for drought indicator variable based on Poisson regression of count on time for exponential model,  $a \cdot \exp(b \cdot \text{year})$ .**

Trend Measure	Unadjusted Proportion		Adjusted Proportion	
Proportion of index areas with estimated <u>decline</u> ( $b < 0$ )	21/24 = 0.88		11/24 = 0.46	
Proportion index areas with <u>significant</u> ( $P < 0.10$ ) <u>decline</u>	10/24 = 0.42		5/24 = 0.21	
Proportion of index areas with estimated <u>growth</u> ( $b > 1$ )	3/24 = 0.125		13/24 = 0.54	
Proportion of index areas with <u>significant</u> ( $P < 0.10$ ) <u>growth</u> (South Fork Salmon River showing significant growth)	1/24 = 0.04		1/24 = 0.54	
Trend Measure	b (% yearly decline)	P	b (% yearly decline)	P
Average of trends	-0.102 (9.7%)	<0.01	-0.053 (5.2%)	0.09
Single trend based on total redd count over index areas	-0.047 (4.6%)	0.28	-0.022 (2.2%)	0.71
Average trends (excluding South Fork Salmon)	-0.114 (10.5%)	<0.01	-0.062 (6.0%)	0.05
Single trend based on total redd count over index areas (exclusing South Fork Salmon)	-0.093 (8.9%)	0.06	-0.064 (6.2%)	0.33

Comparing adjusted and unadjusted trends, every measure indicates less decline associated with the adjustment.

When applied to the recovery period, the climatic adjustment would be expected to have an effect on the trend estimate if climatic conditions improve during the recovery period; that is, the adjusted measures would indicate less growth than the unadjusted measures. Under such conditions, there would be less certainty that a recovery-based increase had occurred (note the lower probability levels associated with the adjusted measures in Table 5-9); and the evaluation period may have to be extended before de-listing could take place.

However, if the climatic conditions vary within the recovery program and there is no overall climatic trend, then adjusting for the covariate would likely decrease the variance around the trend line and thereby reduce the standard error of the trend coefficient. This would lead to a more powerful statistical test, and the recovery program, if it is effective, could be judged as a success in less time than would be the case if the adjustment were not made.

Using the adjustment to assess the overall trend is recommended; however, the adjustment will not be appropriate when assessing extinction probabilities. Climatic variation would likely be a factor that would affect extinction probabilities; therefore its contribution to the variation should not be removed. That is, the covariate adjustment should be made when assessing the magnitude of the trend; however it should not be made when developing extinction models.

We should point out that covariate adjustments are merely statistical adjustments that adjust for linear trend. The nature of trend may not be linear. Further, there is no guarantee that a detected trend with the climatic indicator is actually due to climate; a detected trend may be due to some phenomenon unrelated to time but correlated with the indicator.

**RECOMMENDATION: Trend and abundance measure should be adjusted for environmental factors such as drought, and el Nino events, that are known to affect survival and production but that are not affected by the recovery program.**

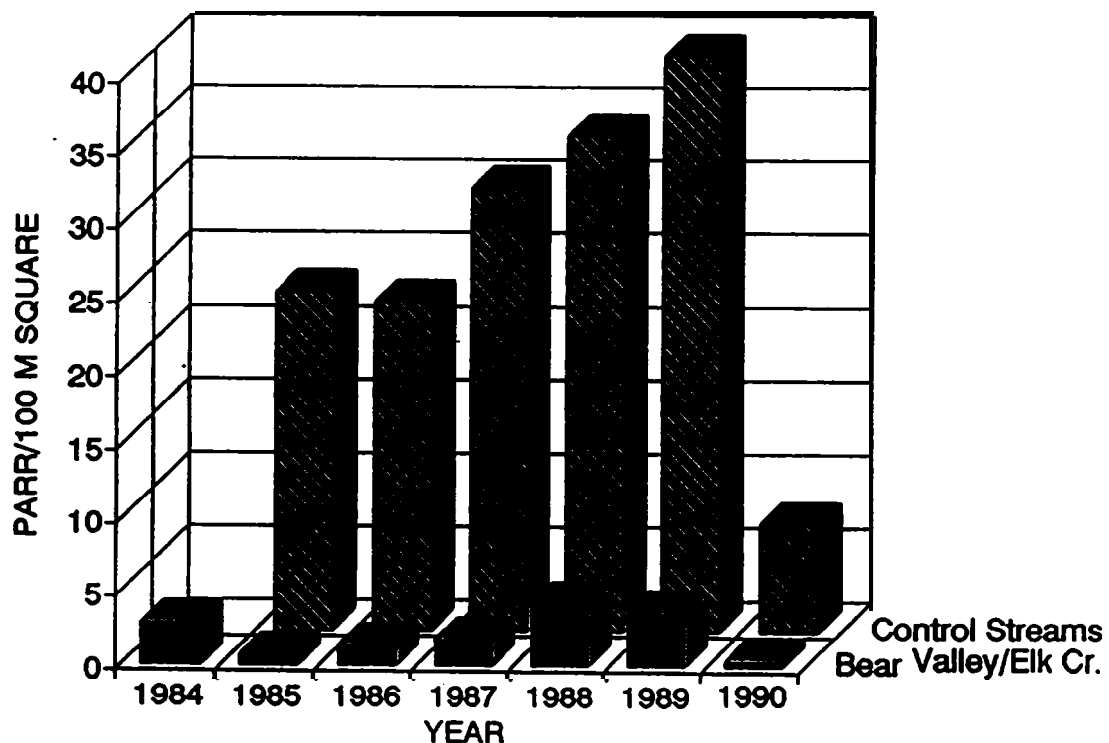
Recovery Target Values. We feel that the target value of the recovery program for redd count should not be the low pre-recovery base from 1986-1990. Rather it should strive to attain a fraction of the seeding capacity of the system. We discuss this under a later section on parr density.

### 5.2.1.3. Parr Density

IDFG has been monitoring parr density in selected streams since 1984.

Parr densities are far below what would be expected based on stream quality. Figure 5-7 (Rich, et al. 1992) presents yearly means of Parr density for two sets of upper Middle Fork Salmon River streams inhabited by wild chinook. Bear Valley Creek and Elk Creek are streams that have suffered considerable habitat degradation and are considered to be in poor conditions whereas the control streams are considered to be pristine streams that are ranked as representing excellent chinook habitat (Bruce Rich, IDFG, Eagle, personal communication). The historical degradation of Bear Valley and Elk Creeks has resulted in reduced production; however all of the streams are considered to be well below their potential carrying capacity. The carrying capacity index used by IDFG is given in Table 5-10.

#### MEAN PARR DENSITIES-OF WILD CHINOOK in established monitoring sections



**Figure 5-7.** Mean parr densities of wild chinook in established monitoring sections (Rich et al. 1992).

**Table 5-10: Expected parr densities for different qualities of stream habitat**

<b>Stream Quality</b>	<b>Expected Parr Density/100 m<sup>2</sup></b>
<b>Poor</b>	<b>12</b>
<b>Fair</b>	<b>44</b>
<b>Good</b>	<b>77</b>
<b>Excellent</b>	<b>108</b>

The parr density geometric means of 1.5 parr/m<sup>2</sup> for the poor quality streams and 22.8 parr/m<sup>2</sup> for the excellent quality streams are only a small fraction of the expected density. Although degradation of habitat may have reduced populations, it is likely that construction of hydro-electric dams have so severely impacted juvenile and adult passage that an insufficient number of spawners are returning to even come close to the expected parr density of the excellent quality streams.

This suggests that the target abundance value of the recovery program should be expressed in terms of seeding capacity of the habitat, not in terms of a specific number of returning fish or number of redds. It may be necessary to relate the number of redds to the parr density and to express the recovery in terms of the redd count, but parr density should also be the focus.

The parr sampling areas have been chosen independent of the redd index area locations. However, three of the surveyed streams (Bear Valley, Elk and Sulfur creeks) are also sites of redd index areas. We correlated the mean parr densities for each of these streams with the previous year's redd count from the respective stream's index area. The correlations are given in Table 5-11.

Table 5-11: Correlation coefficients between parr densities and previous year's redd count (n = 6)	
Stream	Correlation Coefficient
Bear Valley Creek	0.60
Elk Creek	0.82
Sulfur Creek	0.43

Because of the small sample size, only one of the correlation coefficients was found to be significant ( $P < 0.10$ ). We were not able to identify which parr sampling section, if any, within the stream was associated with the redd index area. Were we able to do so, the correlation coefficients may have been higher. Even, so these correlations are high enough to suggest that an integrated strategy for sampling both redds and parr density areas for enumeration might provide a basis for using redd counts, or possibly redd density, as an indirect measure parr density.

**RECOMMENDATION: Develop integrated sampling strategies for parr density and redd count monitoring to permit the recovery target values based on redd counts to take seeding capacity into account.**

#### 5.2.1.4. **Lower Granite Smolt Passage**

##### **Historic Information**

**Smolt Passage Estimates.** From 1975 through 1983, NMFS sampled smolt passage from the bypass. The following expansion was used to estimate of total smolt passage (TSP) of spring-summer chinook. The daily estimate was

$$\text{TSP} = \frac{\text{Count in Sample (c)}}{[\text{Sample rate (r)}][\text{Collection Efficiency (e)}]}$$

The sample rate (r) is the proportion of the bypass passage that was sampled, therefore  $c/r$

is the estimated bypass passage. The collection efficiency (e) was the predicted proportion of fish entering the bypass based on a linear calibration for percent flow through the bypass.

The calibration equation was developed by regressing the estimated proportions of marked releases entering the bypass on the percent flow (f) through the powerhouse on the day of passage, the percent flow being

$$f = \frac{\text{power house flow}}{\text{power house flow} + \text{spill}}$$

The estimated calibration parameters were then used to predict the collection efficiency for each day of sampling. The calibration would be unbiased under the following conditions:

- 1) collection efficiency is linearly related to percent flow through the powerhouse;
- 2) the estimated collection efficiencies for the marked releases used to estimate the calibration equation were unbiasedly adjusted for the release's mortality rate from the point of release to the dam;
- 3) for each evaluated stock, the proportion of fish entering the bypass increased linearly with flow through the powerhouse;
- 4) the probability of released fish passing via the powerhouse was the same as that for each stock being evaluated; and
- 5) the probability of released fish entering the bypass was the same as that for each evaluated stock.

The TSP estimate is not adjusted for the fish guidance efficiency (FGE).

Estimates of wild and hatchery spring chinook passage and summer chinook passage are given in Table 5-12 (Raymond 1988). The estimation procedure was extended to the 1984 passage. 1984 was the year that the Fish Passage Center initiated its sampling program for estimating the fish passage index.

Table 5-12. Estimates of Lower Granite Dam total smolt passage (TSP), 1975-1984, and of Lower Granite Dam smolt passage index (SPI), 1984-1992.

Year	NMFS's Total Smolt Passage Estimates*				Smolt Passage Index (SPI) Estimates**				
	Spring Chinook (hatchery)	Spring Chinook (wild)	Summer Chinook	Total Spring/summer	SPI Sub-yearling Chinook	SPI Yearling Chinook	SPI Total Chinook	Chinook Hatchery Releases	SPI to Hatchery Ratio
1975	2.20	1.70	0.50	4.40					
1978	2.40	1.90	0.60	4.90					
1977	1.20	0.60	0.20	2.00					
1978	2.00	0.70	0.30	3.00					
1979	2.30	1.30	0.50	4.10					
1980	2.40	2.20	0.60	5.20					
1981	2.30	0.60	0.40	3.30					
1982	1.40	0.20	0.40	2.00					
1983	2.60	0.80	0.40	3.80					
1984	4.20	0.70	0.50	5.40					
1985							1,245,400	8,838,000	0.14
1986							1,812,000	7,997,000	0.23
1987							1,700,300	6,496,000	0.26
1988							2,499,000	11,708,000	0.21
1989							2,798,900	11,427,000	0.24
1990							2,583,000	11,477,000	0.23
1991					13,900	2,295,700	3,199,600	12,488,000	0.26
***1992					5,943	2,500,719	2,309,600	9,767,000	0.24
							2,506,662	10,900,000	0.23

\* Raymond (1988)

\*\* Fish Passage Center, Annual Reports (1989-1991)

\*\*\* Hatchery releases from T. Bergren (Fish Passage Center, personal communication)

**Fish Passage Index.** From 1984 to the present, the Fish Passage Center estimates a smolt passage index (SPI) that is not, and is not intended to be, an estimate of total smolt passage. It is effectively a measure of bypass passage adjusted for flow through the powerhouse. The daily estimate

$$\text{SPI} = \frac{\text{Count in Sample (c)}}{[\text{Sample rate (r)}][\text{Proportion Flow through Powerhouse (f)}]}$$

The division by the proportion of the flow diverted through the powerhouse is intended to be an adjustment for different daily operations (spill and unit loading). If there is no spill, SPI becomes c/r, the estimate of actual bypass passage.

The fish passage index will be linearly related to total passage under the following conditions:

- ▶ for each evaluated stock, fish pass via spill and powerhouse units in numbers proportional to the flow through these passage routes, and
- ↗ for each evaluated stock, the collection efficiency is independent of percent flow through the powerhouse.

This index does not attempt to adjust for the fish guidance efficiency (FGE). The FGE differs among species and dams, therefore the smolt passage indices are not comparable among dams or among species. The index would be comparable over years at a given dam provided the FGE remains stable across years. (T. Berggren, Fish Passage Center, Portland.)

Through 1992 the index was not partitioned into wild and hatchery components, and through 1990 the index was not partitioned into yearling (age 2) and sub-yearling (age 1) categories; therefore 1991 and 1992 are the only years to date for which there are separate estimates of what would be taken to be spring-summer and fall chinook. Estimates of the SPI are given for chinook in Table 5-12.

**Evaluation of Historical Data.** We made no attempt to merge the 1975-1983 TSP estimates with the 1985-1992 SPI estimates. Although there was a potential calibration year, 1984, in which both estimates were available, they differed dramatically in ways that precluded the common year being a base of calibration. The NMFS estimated total passage to be 5.4 million total chinook (excluding fall chinook), and the SPI chinook estimate was 1.25 million. The more than four-fold increase of NMFS abundance over the SPI is unlikely to reflect reality. Snake River runoff in 1984 was well above normal and spill at Lower Granite Dam frequently exceeded 50% of flow from mid-April through mid-June (1984 Annual Report from Water Budget Center to BPA). However, the estimated TSP exceeded those for all previous years, whereas the SPI estimate was lower than those for all subsequent years.



Even accounting for the fact that the TSP is intended to be an abundance measure whereas the SPI is only an index, there is an indication that at least one of the measures is biased: The estimated TSP exceeded the TSPs from all previous years, whereas the SPI estimate was lower than the SPIs for all subsequent years. For years subsequent to 1984, the ratio of SPI to total hatchery smolt released above Lower Granite Dam was quite stable, ranging from 0.21 to 0.26. If the proportion of hatchery smolts comprising the run was fairly constant over that period, then the ratio indicates that the SPI would be a reasonable index measure. However, in 1984 that ratio was substantially lower (0.14). If the proportion of the run comprised of hatchery fish was comparable to subsequent years, then it would appear that the SPI was a biased indicator in 1984. We were not able to compute comparable smolt release numbers for years prior to 1984; therefore we could not determine whether there was an indication of bias in the TSP estimate.

It seems inappropriate to use the 1984 overlap between the NMFS's TSP estimate and the Fish Passage Center's SPI to calibrate smolt passage for assessing a single trend over the whole time period. Were such a calibration possible, it would be biased because the TSP estimate did not include fall chinook whereas the SPI included all chinook stock. Therefore, separate estimates of trend were made for TSP and SPI estimates. A simple linear regression of the log of the TSP estimate on year gave an estimated 11% per year reduction in wild fish between 1975 and 1984; the estimate was not significant ( $P=0.14$ ).

No wild assessment could be made based on the SPI because it was not possible to separate the passage into wild and hatchery proportions.

### **Future Smolt Monitoring**

**Smolt Passage Index.** In order to assess whether the recovery program is improving smolt-to-adult survival to Lower Granite, a survival measure of the form

$$S(\text{smolt-to-adult}) = \frac{\text{LGR Wild Adult Count (t)}}{\text{LGR Wild Smolt Count (t-}\Delta\text{t)}}$$

will have to be assessed.

(It would also be desirable to estimate the wild smolt production as a function of parental wild returns two years previously, but this would not be measurable at the dam because it would not be possible to proportionally allocate natural smolt to natural-origin spawners.

We regard the allocation of number of redds to wild spawners based on the proportion of wild spawned out carcasses as the most easily obtainable measure of wild fecundity and success.)

Estimating S(smolt-to-adult) requires a suitable measure of smolt abundance. Would the SPI be a suitable indicator of survival? If SPI were highly correlated with smolt passage, then the ratio

$$R(\text{smolt-to-adult}) = \frac{\text{LGR Wild Adult Count (t)}}{\text{SPI}(t-\Delta t)}$$

should be a good indicator of trend in smolt-to-adult survival.

Beginning in 1993 the SPI will be estimated separately for hatchery and wild fish. The stability of the SPI from 1985-1992 as indicated by the ratio

$$r = \text{SPI} / (\text{Total hatchery smolt release})$$

was promising. The suitability of  $r$  as a stability measure was contingent on a relatively constant proportion of the passage being hatchery products.

For the year, 1984, in which the measure  $r$  was not consistent, one might have attempted to adjust the inconsistent SPI for the consistent ratio.

$$\text{SPI}(\text{adj}) = \frac{r(\text{cons})}{r(\text{incons})} * \text{SPI}(\text{incons})$$

adj = adjusted  
cons = consistent  
incons = inconsistent

however, such an adjustment would have to be based on the assumption that the proportion of hatchery smolts in the run was the same for the consistent and inconsistent period.

With all hatchery fish being marked and all marked fish being separately enumerated, it should be possible to adjust  $r$  over time for the proportion of hatchery fish. It seems reasonable that  $r$  would increase with hatchery proportion,  $p(\text{hat})$

$$r = b \cdot p(\text{hat})$$

the proportion hatchery can be estimated from the count. Then, if conditions existed that rendered the SPI inconsistent with most of the record, it would be possible to make an adjustment using

$$r(\text{adj}) = b \cdot p(\text{hat}')$$

$p(\text{hat}')$  being the proportion hatchery estimated for the inconsistent SPI.

We advise against pooling smolt passage indices over Snake River dams for two reasons;

- Unless all smolts are collected out of the bypass of a dam, there is a probability that they will be counted at other downstream dams.
- Smolt passage indices are not comparable over dams because of differences in the FGEs.

If there is a desire to exploit SPI information from all Snake River dams, then the use of multi-variate analysis techniques (such as canonical correlation or principle components) should be explored.

**Trucked and Barged Smolts.** If all smolts are trucked and barged, the total count of transported fish could be used to estimate smolt passage through the bypass. It would be possible to total the number of transported fish over dams. It would be necessary to estimate the wild proportion of the passage, and to multiply the transported number by that proportion. This proportion could be estimated from the daily SPI hatchery and wild counts. Another alternative would be to pit-tag a known proportion of all hatchery releases. The number of pit-tags read at the bypass could be divided by that proportion to obtain an estimate of the hatchery production. However, pit-tagging might prove to be a costly undertaking.

Even if transported numbers are used as the smolt measure the SPI should still be used as an indicator in case there are times or years when there is no transportation.

#### **5.2.1.5 Demographic Characteristics**

The decision to consider the Snake River spring and summer chinook as a single evolutionary significant unit was based primarily on protein electrophoresis, presumably reflecting the allelic distribution of neutral genes (Matthews and Waples 1997). However, run time is a genetically heritable characteristic that may permit a maximum exploitation of the habitat. The recovery program should strive to maintain the temporal and spacial distribution of the spring-summer chinook to guarantee the preservation of genetic traits which permit the stock to optimally exploit its habitat.

#### **Shift in Run Time**

The summer proportion of the run has decreased over time. As is evidenced in Figure 5-8 (based on dam counts given in Appendix 3), the summer passage at Ice Harbor Dam comprised 48 percent of the adult run from 1962 through 1966 but declined to 23 percent for the fifteen year period, 1977-1991. It is not possible to tell whether this change in the run composition is due to a natural decline or due to human activities such as harvest, dam operation, or a heavier hatchery production of the spring portion of the run. The proportion of the summer wild redd count to total wild redd count (based on wild spring and summer counts presented in Appendix 6) in the Salmon River subbasin does not show a trend over time.

**RECOMMENDATION: The recovery program should try to ascertain whether the shift in the run is due to human activities. It should take action to guarantee that the summer portion of the run does not decline further, and it should consider possible actions that would lead to the reestablishment of the historic run distribution.**

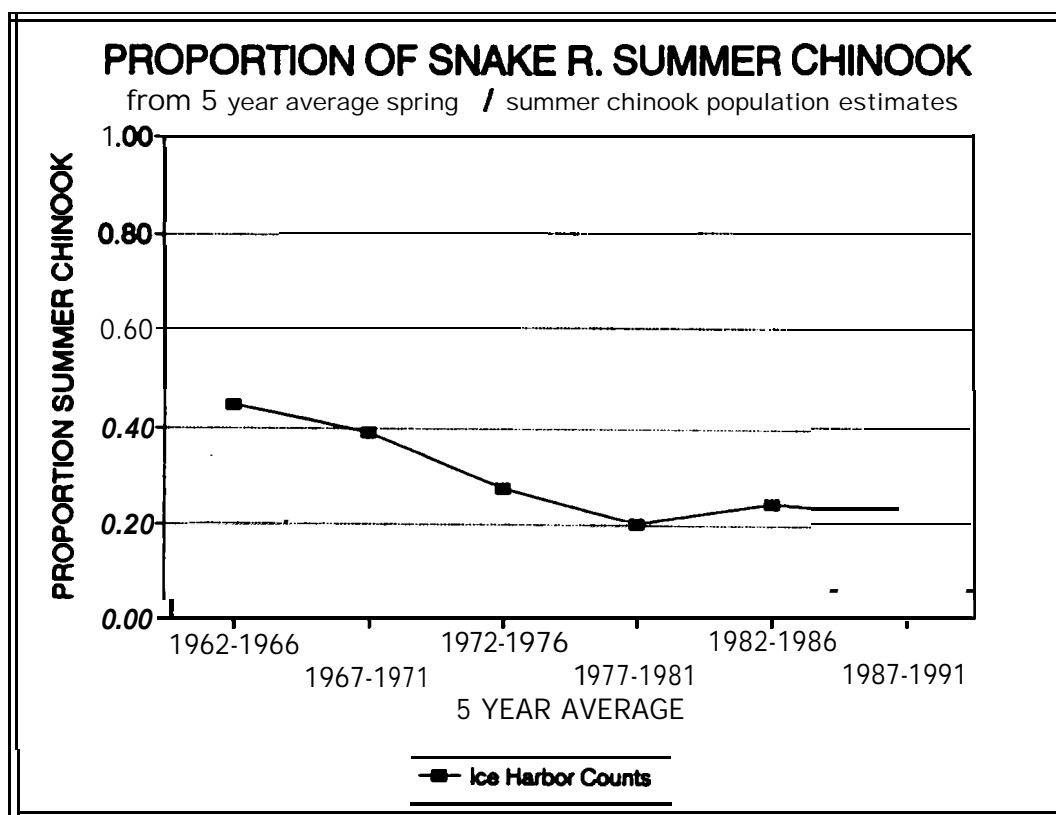


Figure 5-8. Summer proportion of Snake River spring-summer chinook.

### Age Distribution

We have not been able to obtain a continuous record of spawning ground surveys for age distribution. The longest continuous record that we have from ODFW for the Imnaha and Grande Ronde subbasins is from 1961 through 1975 (Oregon Fish Commission Reports, Northeast Oregon Spawning Ground Surveys), we have been able to supplement this with records covering 1986 through 1992 (Paul Hirose, ODFW, personal communication). In some cases we were able to get information on the relative frequencies of the age classes but not on the actual count.

For the Salmon River and Clearwater subbasins, we went back through IDFG's spawning ground survey reports until 1970. There were some gaps in the record, but we were able to obtain a consistent record for the years 1970 and 1972 through 1981 and obtain some supplemental records from 1985 through 1988 (IDFG Salmon Spawning Ground Surveys).

Figure 5-9 presents the age-distributions of dead fish pooled over the years 1970 through 1975 for the Imnaha and Grande Ronde index areas. Figure 5-10 presents the size distribution of spawned-out carcasses pooled over those same years for the Salmon River index areas. 1970 through 1975 represents the common period between the ODFW and IDFG records. The data used to generate the figures are presented in Table 5-13 and Table 5-14 which give additional years' information. Some of the records gave only the relative age-class distributions, but not the total redd count; therefore the means presented are not based on the whole data set. The figures should only be interpreted in conjunction with the table data.

Age distribution tends to be highly variable because of variable production over brood years. Even so, the age distribution appears to vary within the subbasins as well as among the subbasins.

In the Grande Ronde subbasin, Minam River, Looking Glass Creek, and Catherine Creek are dominated by age 4 adults, and this distribution has been reasonably consistent until recent years when hatchery strays may have been influencing the carcass counts. Lostine River also tends toward age 4 adults but is far more variable over time. Bjorn et al. (1991) in a study of radio-tagged spring- and summer-run components of the run found that both components of the run enter the Grande Ronde with more springs than summers but that all fish returning to Lookingglass Hatchery were marked during the spring portion of the run. All Grande Ronde redds are currently allocated to the spring component of the run in reconstructed LGR wild counts.

Although Figure 5-9 indicates that Imnaha subbasin carcasses tends to be more dominated by age-five fish, when viewed over all years the distribution between the two age classes tends to be even but highly variable over time. The Imnaha has been designated as a spring component in reconstructed LGR counts. Bjorn et al. found equal portions of the summer- and spring-tagged fish in the river.

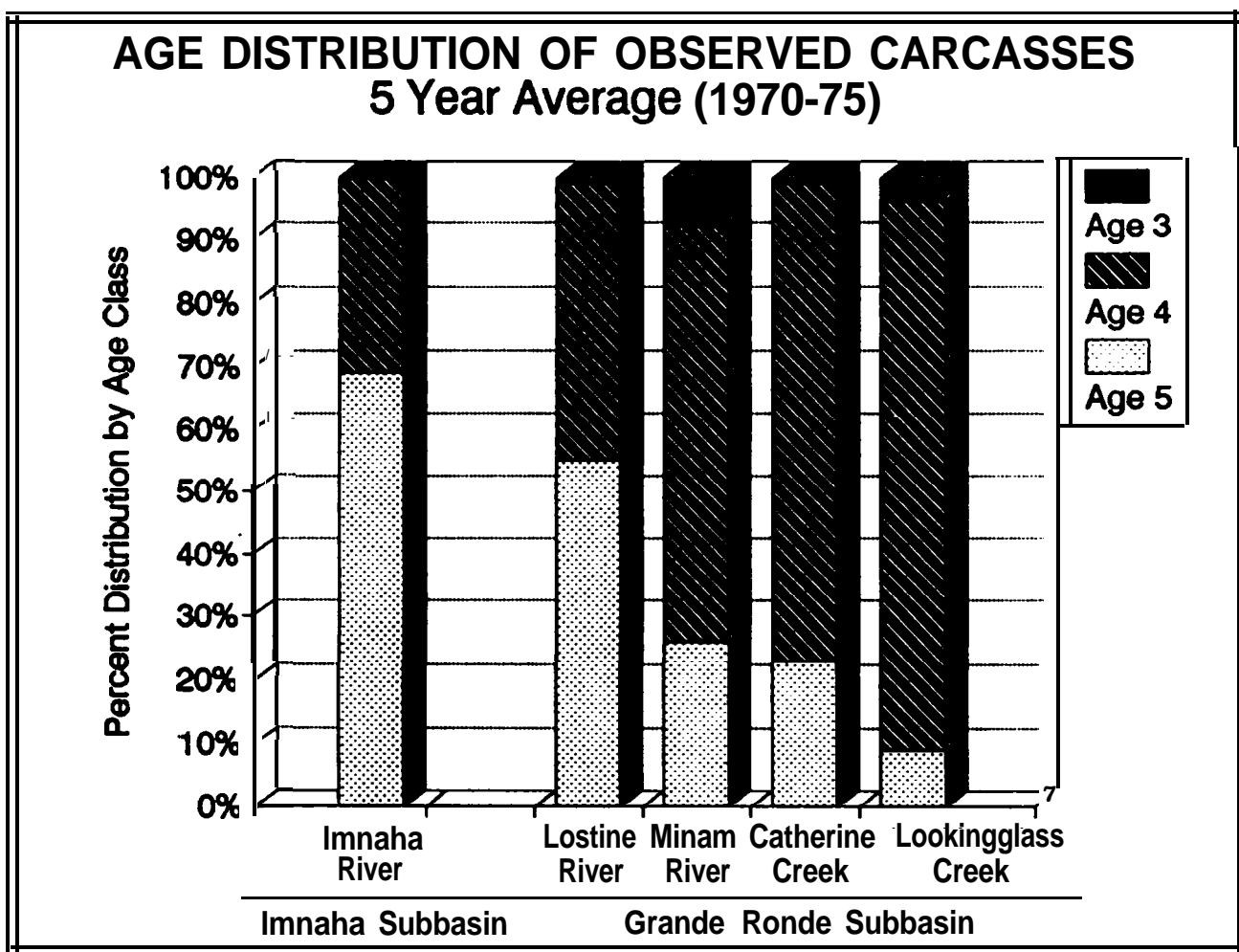


Figure 5-9. Age distributions of observed carcasses, Imnaha and Grande Ronde subbasins, pooled 5-year average (1970, 1972-1975).

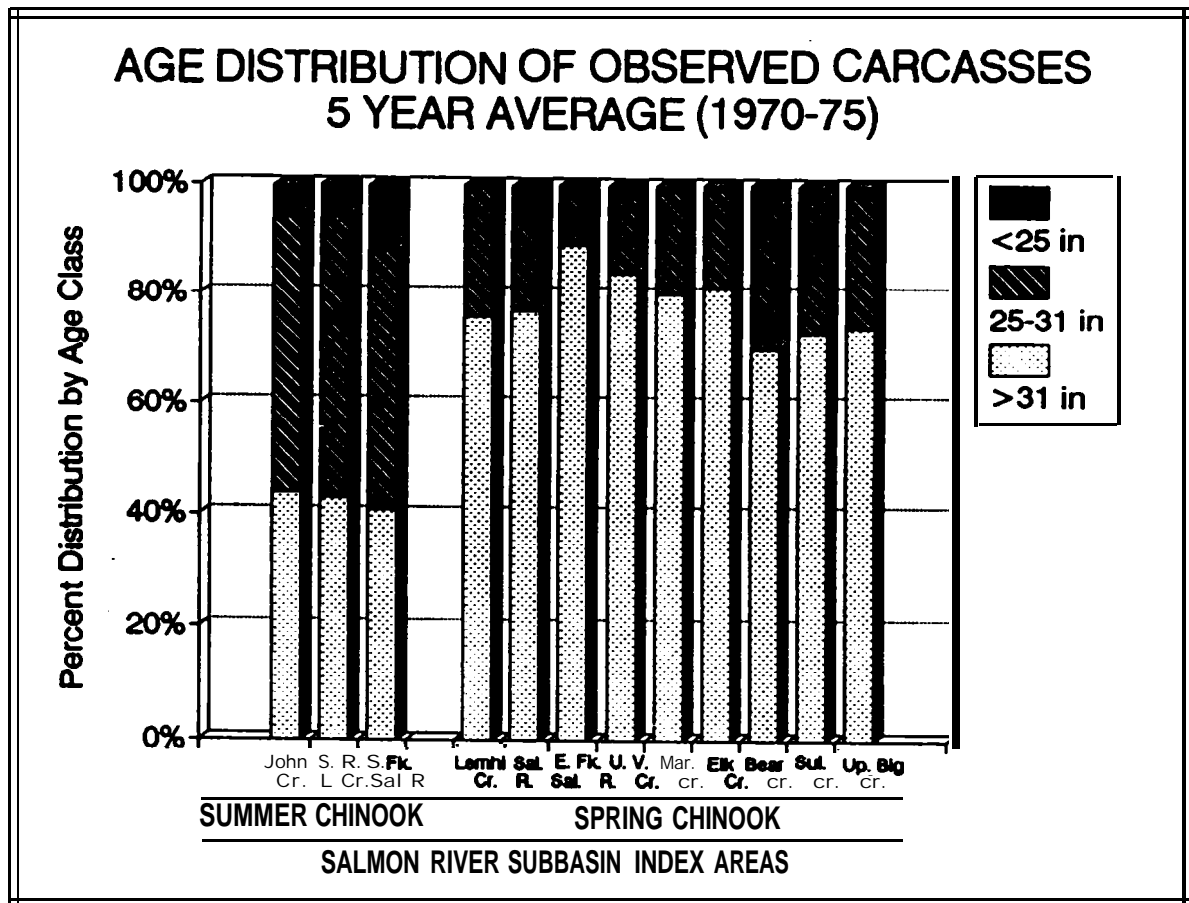


Figure 5-10. Fork length distributions of observed carcasses, Salmon River subbasin, pooled 5-year average (1970, 1972-1975).



Table 5-13. Age distribution of carcasses in Imnaha and Grande Ronde subbasins.

	Imnaha Subbasin					Grande Ronde Subbasin																					
	Imnaha River					Lostine River					Minam River				Catharine Creek					Lockingtons Creek							
	Year	Age 3	Age 4	Age 5	Age 6	Count	Age 3	Age 4	Age 5	Age 6	Count	Age 3	Age 4	Age 5	Count	Age 3	Age 3	Age 4	Age 5	Count	Age 3	Age 4	Age 5	Count			
	1991	0.14	0.02	0.04	0.00	7	0.15	0.05	0.05	0.00	7	0.05	0.00	0.17	7	—	—	—	—	—	0.00	100.00	0.00	7			
	1992	0.05	0.07	0.05	0.00	60	0.20	0.05	0.15	0.00	20	0.11	0.04	0.25	25	0.00	0.00	1.00	0.00	15	0.00	0.00	0.10	10			
	1993	0.16	0.02	0.00	0.00	24	0.10	0.05	0.07	0.00	20	0.00	0.00	0.40	15	—	—	—	—	—	0.00	0.00	0.00	2			
	1994	0.05	0.05	0.05	0.01	105	0.05	0.75	0.15	0.00	117	0.05	0.05	0.15	45	—	—	—	—	—	0.17	0.77	0.05	25			
	1995	0.10	0.00	0.00	0.00	10	0.05	0.04	0.27	0.00	11	0.00	0.05	0.17	5	0.00	0.05	0.07	0.00	5	0.12	0.05	0.02	7			
	1996	0.13	0.05	0.02	0.00	85	0.15	0.00	0.25	0.01	134	0.00	0.05	0.45	11	0.00	0.00	0.75	0.05	4	0.05	0.05	0.01	7			
	1997	0.05	0.04	0.01	0.01	70	0.11	0.45	0.42	0.01	75	0.05	0.07	0.25	12	0.00	0.00	0.75	0.05	4	0.21	0.05	0.05	7			
	1998	0.01	0.05	0.05	0.01	55	0.05	0.00	0.45	0.00	44	0.05	0.71	0.05	31	0.00	0.00	1.00	0.00	5	0.47	0.01	0.02	7			
	1999	0.04	0.05	0.05	0.00	75	0.05	0.01	0.25	0.00	70	0.07	0.02	0.11	45	0.05	0.05	0.04	0.00	31	0.04	0.75	0.01	7			
	1970	0.00	0.41	0.57	0.02	45	0.00	0.77	0.05	0.00	20	0.12	0.05	0.15	25	0.00	0.00	0.05	0.11	9	0.05	0.05	0.05	7			
	1971	0.10	0.05	0.00	0.01	80	0.15	0.04	0.15	0.00	25	0.15	0.00	0.27	15	0.00	0.05	0.05	0.00	19	0.15	0.05	0.04	7			
	1972	0.01	0.05	0.75	0.00	80	0.05	0.15	0.00	0.00	40	0.05	0.75	0.14	25	0.00	0.05	0.75	0.05	40	0.15	0.75	0.05	7			
	1973	0.00	0.04	0.75	0.00	75	0.00	0.45	0.04	0.00	45	0.07	0.47	0.47	15	0.00	0.00	0.05	0.17	12	0.05	0.05	0.05	257			
	1974	0.00	0.44	0.05	0.05	77	0.00	0.00	0.00	0.00	22	0.05	0.05	0.05	15	0.00	0.00	0.74	0.05	31	0.05	0.05	0.07	75			
	1975	0.00	0.15	0.01	0.00	51	0.00	0.05	0.04	0.00	14	0.00	0.00	0.00	5	0.00	0.00	0.00	1.00	1	0.00	0.05	0.07	5			
	1991-1975 pooled	0.05	0.44	0.50	0.01	942	0.05	0.57	0.25	0.00	254	0.11	0.05	0.21	210	0.01	0.05	0.04	0.15	175	0.05	0.05	0.05	255			
Years comparable to Salmon River Subbasin																											
	1970-75 pooled*	0.00	0.31	0.05	0.01	205	0.01	0.44	0.05	0.00	154	0.05	0.05	0.25	55	0.00	0.01	0.75	0.05	55	0.04	0.07	0.05	245			
	1995	0.05	0.44	0.45	0.00	25	0.05	0.47	0.47	0.00	17	0.00	0.01	0.05	15	0.00	0.00	1.00	0.00	5	0.00	1.00	0.00	2			
	1997	0.00	0.05	0.05	0.00	25	0.00	0.00	0.40	0.00	25	0.00	0.05	0.14	43	0.00	0.05	0.05	0.05	54	0.00	1.00	0.00	2			
	1998	0.05	0.17	0.00	0.00	125	0.00	0.05	0.75	0.00	75	0.05	0.05	0.01	23	0.00	0.05	0.05	0.45	125	0.00	0.05	0.04	25			
	1999	0.17	0.05	0.05	0.05	25	0.05	0.75	0.25	0.00	25	0.00	0.00	0.00	5	0.00	0.00	0.00	0.10	10	0.00	0.75	0.02	5			
	1990	0.05	0.05	0.10	0.00	32	0.00	0.00	0.40	0.00	15	0.00	0.75	0.01	15	0.00	0.00	0.75	0.05	5	0.00	1.00	0.00	2			
	1991	0.07	0.05	0.04	0.00	51	0.00	0.15	0.04	0.00	25	0.07	0.25	0.07	14	0.00	0.05	0.01	0.15	15	0.00	0.00	0.00	10			
	1992	0.05	0.00	0.17	0.00	155	0.00	0.75	0.27	0.00	20	0.00	0.00	0.00	45	0.00	0.00	1.00	0.00	5	0.00	0.00	0.00	2			
	1995-1999 pooled	0.05	0.05	0.40	0.00	482	0.01	0.45	0.54	0.00	227	0.01	0.05	0.21	151	0.00	0.05	0.70	0.05	250	0.00	0.45	0.04	45			

\* - Excluding 1971

Sources: 1991-1975 data from: Bennett, Donald E. (Fish Commission of Oregon, Clatskanie, Or.) 1975, "Northeastern Oregon Spring Chinook Spawning Ground Surveys for 1975"  
1995-1999 data from: Paul Hirose (ODFW) personal communication.

Table 5-14. Fork length distributions of carcasses for index areas in Salmon River subbasin

	SUMMER CHINOOK INDEX AREA											SPRING CHINOOK INDEX AREA			
	Johnson Creek			Seossh River, Lake Creek				South Fork Salmon River				Lemhi River			
	Year	<25in	25-31in	>31in	<25in	25-31in	>31in	Count	<25in	25-31in	>31in	Count	<25in	25-31in	>31in
1970	0.23	0.40	0.38	0.29	0.46	0.25	24	0.18	0.52	0.31	885	0.01	0.34	0.64	87
1972	0.04	0.63	0.34	0.03	0.54	0.43	81	0.11	0.56	0.31	282	0.00	0.27	0.73	181
1973	0.04	0.35	0.60	0.00	0.43	0.57	37	0.08	0.38	0.54	604	0.00	0.16	0.84	129
1974	0.03	0.55	0.41	0.00	0.74	0.25	4	0.08	0.55	0.37	156	0.03	0.18	0.79	33
1975	0.00	0.16	0.84	0.00	1.00	0.00	1	0.10	0.41	0.49	182	0.00	0.24	0.76	29
1976	0.06	0.36	0.58	0.20	0.60	0.20	5	0.46	0.21	0.33	24	0.00	0.50	0.50	14
1977	0.03	0.88	0.12	0.00	0.82	0.08	37	0.04	0.84	0.12	386	0.13	0.50	0.38	16
1978	0.00	0.08	0.91	0.00	0.03	0.97	37	0.00	0.12	0.88	404	0.00	0.06	0.95	74
1979	0.20	0.10	0.70	—	—	—	—	0.31	0.08	0.63	16	0.00	0.00	1.00	4
1980	0.10	0.67	0.24	0.50	0.50	0.00	4	0.14	0.57	0.29	7	—	—	—	—
1981	0.08	0.33	0.59	0.07	0.71	0.21	14	0.17	0.50	0.33	6	0.00	0.57	0.43	7
1970-1981 pooled	0.05	0.50	0.45	0.08	0.51	0.43	254	0.10	0.47	0.44	2882	0.01	0.23	0.76	954
Excluding 1977	0.05	0.44	0.51	0.07	0.44	0.49	217	0.11	0.41	0.49	2296	0.01	0.22	0.77	538
Years Comparable to Imnaha and Grande Ronde Subbasins															
1970-1975 pooled	0.06	0.50	0.44	0.06	0.51	0.43	157	0.12	0.47	0.41	1636	0.00	0.24	0.75	439
1985				0.04	0.62	0.35	26	0.14	0.39	0.47	113				
1986				0.04	0.59	0.37	27	0.09	0.25	0.66	84				
1987				0.03	0.58	0.39	74	0.03	0.65	0.32	117				
1988				0.04	0.16	0.80	119	0.03	0.09	0.89	433				
1985-1988 pooled				0.04	0.38	0.58	246	0.05	0.24	0.71	727				

\* In 1977 summer chinook carcass counts were male dominated:

96% males on Johnson Creek  
 95% males on Seosch Creek  
 85% males on South Fork

Source: IDFG Spawning Ground Surveys

Table 5-14. Continued

SPRING CHINOOK INDEX AREA																
Year	Salmon River				East Fork Salmon				Upper Valley Creek				Marsh Creek			
	<25in	25-31in	>31in	Count	<25in	25-31in	>31in	Count	<25in	25-31in	>31in	Count	<25in	25-31in	>31in	Count
1970	0.19	0.34	0.47	132	0.08	1.19	0.75	229	0.07	0.33	0.80	18	0.01	1.30	0.88	120
1972	0.10	0.16	0.72	289	0.01	1.14	0.86	151	0.05	0.22	0.74	65	0.02	1.36	0.72	47
1973	0.04	0.12	0.86	252	0.00	1.01	0.89	298	0.00	0.04	0.86	89	0.01	1.08	0.92	120
1974	0.20	0.27	0.73	77	0.06	1.14	0.81	57	0.07	0.30	0.83	43	0.00	1.41	0.59	39
1975	0.03	0.12	0.86	488	0.00	1.06	0.86	81	0.00	0.08	0.82	89	0.02	1.09	0.89	98
1976	0.10	0.27	0.63	97	0.00	1.00	1.00	4	—	—	—	—	0.00	1.20	0.40	5
1977	0.16	0.46	0.39	743	0.02	1.09	0.89	81	—	—	—	—	0.00	1.03	0.32	25
1978	0.00	0.02	0.86	780	0.00	1.02	0.19	432	0.00	0.00	1.00	34	0.00	1.03	0.97	206
1979	0.05	0.05	0.89	96	0.07	1.00	0.03	6	0.00	0.00	1.00	1	0.05	1.03	0.75	4
1980	—	—	—	—	-0-	-0-	-0-	—	—	—	—	—	-0-	-0-	—	—
1981	0.06	0.73	0.21	52	0.00	1.06	0.05	8	—	—	—	0	0.00	1.03	0.80	5
1970-1981 pooled	0.07	0.21	0.72	2927	0.01	0.7	0.82	1446	0.02	0.13	0.86	236	0.01	0.16	0.82	886
Excluding 1977	0.04	0.12	0.83	2164	0.01	0.7	0.82	1395	0.02	0.13	0.86	236	0.01	0.14	0.84	843
Years Comparable to Ingham and Grande Ronde Subbasins																
1970-1975 pooled	0.06	0.17	0.77	21	0.00	0.08	0.89	915	0.02	0.14	0.83	201	0.01	0.19	0.79	424
1986																
1986																
1987																
1988																
1986-1988 pooled																

Table 5-14. Continued

SPRING CHINOOK INDEX AREA																
Year	Elk Creek				Bear Valley				Sulphur Creek				Upper Big Creek			
	<25in	25-31in	>31in	Count	<25in	25-31in	>31in	Count	<25in	25-31in	>31in	Count	<25in	25-31in	>31in	Count
1970	0.00	0.34	0.77	300	0.06	0.30	0.65	286	0.09	0.29	0.62	34	0.05	0.19	0.76	38
1972	0.02	0.44	0.54	170	0.06	0.44	0.51	97	0.00	0.41	0.59	17	0.00	0.23	0.77	13
1973	0.01	0.04	0.96	480	0.02	0.09	0.89	234	0.01	0.22	0.77	103	—	—	—	—
1974	0.00	0.44	0.56	32	0.03	0.33	0.64	172	0.00	0.17	0.83	8	0.00	0.44	0.56	27
1975	0.00	0.18	0.84	19	0.00	0.13	0.87	30	0.00	0.00	1.00	3	0.00	0.06	0.94	17
1976	0.00	0.29	0.71	7	0.13	0.25	0.63	8	0.00	0.00	1.00	3				
1977	0.01	0.78	0.20	74	0.02	0.79	0.19	118	0.00	1.00	0.00	2	0.00	0.88	0.14	7
1978	0.00	0.04	0.96	131	0.00	0.03	0.96	237	0.00	0.00	1.00	6	0.00	0.00	1.00	38
1979	0.00	0.10	0.80	10	0.00	0.09	0.91	11	0.00	0.00	1.00	2	0.00	0.33	0.67	6
1980					0.00	0.33	0.67	12		—	—		0.00	0.00	1.00	2
1981	0.00	0.88	0.14	7	0.00	0.38	0.63	8	0.00		0.00	1				0
1970-1981 pooled	0.01	0.21	0.79	1100	0.03	0.26	0.71	1213	0.02	0.25	0.73	177	0.01	0.21	0.79	148
Excluding 1977	0.01	0.17	0.83	1026	0.03	0.30	0.77	1085	0.02	0.24	0.74	175	0.01	0.18	0.81	141
Years Comparable to Imnaha and Grande Ronde Subbasins																
1970-1979 pooled	0.01	0.18	0.81	871	0.04	0.26	0.71	819	0.02	0.25	0.72	183	0.02	0.24	0.74	96
													Elk, Bear Valley, Sulphur, Upper Big Creeks Combined			
1985													0.00	0.50	0.50	4
1986													0.04	0.31	0.65	156
1987													0.01	0.28	0.71	110
1988													0.00	0.07	0.92	310
1985-1988 pooled													0.01	0.18	0.80	580

Within the Salmon River subbasin, the summer chinook index areas tend to have approximately equal but highly variable numbers of age 4 and age 5 carcasses, whereas age-five spawners dominate the spring chinook index area carcass counts. The summer chinook's age distribution varies greatly over years; whereas the spring chinook's age distribution remains relatively stable.

In 1977 there was a notable exception to an age 5 domination of the spring chinook distribution. The proportion of age-four carcasses exceeded that of age-five in seven out of eight of the spring index areas that were assessed, the East Fork of the Salmon River being the exception. In that year the summer chinook's normal age-four frequency was far greater than in other years. The reason for these abnormal age distributions was that the male proportion was unusually high, and we discuss this in the next section.

Data sets from the IDFG and ODFW are not comparable. The years for which we have complete records differ between the two agencies and only overlap in five of the six years from 1970 and 1975, the period presented in the figures. ODFW presents the actual age distribution, whereas IDFG presents the distribution by fork-length. IDFG does not analyze its own fish scales for age; it sends its scale samples to ODFW for age determination. The classification by size was not always consistent among the IDFG's spawning ground survey reports. We standardized the reports according to the most prevalent classification:

less than 25 cm - Age 3  
25 cm - 31 cm - Age 4  
more than 31 cm - Age 5

Keeping these differences in mind, comparisons across figures and tables there are major differences between the Grande Ronde and Salmon River subbasins. The chinook of the Grande Ronde are predominately age 4 fish whereas the spring chinook of the Salmon River are predominantly age 5 fish. The Salmon River summers and Imnaha chinook tend to have an equal but highly variable distributions of age 4 and 5 fish.

To what extent these differences are genetic is not known. But the recovery program should direct its evaluation to each subbasin and to each major group having distinct age distributions to protect against losing genes that may provide the group with characteristics best suited for its environment.

There are size by age overlaps for each age group as illustrated in Table 5-15 which is summarized from ODFW spawning ground surveys from 1961 through 1975.

Table 5-15. Size distribution of different age Spring Chinook carcasses from Imnaha and Grande Ronde subbasins (1962-1964). Data are the proportion of the length interval that was composed by the given age of fish.			
Age	Size		
	less than 25 cm	25 to 31 cm	more than 32 cm
Age 3	0.99	0.01	0.00
Age 4	0.10	0.78	0.11
Age 5	0.00	0.23	0.76

Size distribution by sex is provided in the ODFW reports, but the age distribution by sex is not, nor is it in IDFG reports. Such information would be valuable in understanding the population dynamics of the stock. The fact that males tend to return at a younger age than females may be an adaptive trait that guarantees a certain degree of gene exchange among brood-years.

**RECOMMENDATION: The same age classification procedures should be followed over agencies. Specifically, we recommend that the actual age of the fish be tallied, not fork length classification. Age classifications should be reported separately for each sex.**

The age distribution is also important in assessing the recovery trend. As discussed in Chapter 4, the log ratio,  $\ln\{y(t)/[y(t-\Delta t)]\}$  would be the most biologically appropriate measure to assess an exponential growth trend. When applied to redd counts, the brood year denominator should be a weighting of the brood years that contribute to the return year numerator.

$$y(t-\Delta t) = w(3)*y(t-3) + w(4)*y(t-4) + w(5)*y(t-5)$$

where, for example,  $w(4)$  is the proportion of age-four brood-year redds that contributed as spawners to redds in year  $t$ . The best way to estimate the weights is from the spawning ground surveys.

From the analysis, there appears to be at least four groupings of index redds that should be assessed separately with respect to age allocation:

- ▶ Imnaha subbasin redds
- ▶ Grande Ronde subbasin redds
- ▶ Salmon River subbasin spring redds
- ▶ Salmon River subbasin summer redds

We were not able to assess the Clearwater subbasin for sex distribution of returning adults.

**RECOMMENDATION: Success of the recovery program should be evaluated separately for summer and spring components within each subbasin.**

As we mentioned in the previous section, to determine whether the recovery program is effective in restoring the natural population it will be necessary to separately enumerate hatchery spawners and natural spawners in areas of hatchery influence and to allocate the redds according to those proportions. This will be somewhat artificial since natural and hatchery fish may mate with each other, but the allocation will reflect the contribution of the natural spawners. Such an allocation should be based on spawned-out carcasses.

**RECOMMENDATION: All dead fish are currently counted. A separate tally should be given for hatchery and natural spawned-out female carcasses to permit allocation of redds to natural spawners.**

### **Sex Distribution**

We have a reasonably extensive data set on sex distribution of carcasses only for the Salmon River subbasin. The sex ratio varies dramatically over years. However, the relative frequency of females is greater for spring chinook than for summer chinook (Table 5-16). The age distribution may be an important demographic characteristic in its own right. But it is also used to estimate the number of spawners. The standard procedure for estimating the number of spawners is to multiply the number of redds by the inverse of female frequency.

$$\text{Spawners} = \{1/[\text{female proportion of spawners}]\} \cdot \text{redds}$$

If the sex ratio is one-to-one, then female proportion is 0.5, in which case the expansion is 2.

Table 5-16. Sex distribution from spawning ground surveys, Salmon River subbasin.

Year	SUMMER CHINOOK INDEX AREA						SPRING CHINOOK INDEX AREA					
	Johnson Creek			Secesh River, Lake Creek			South Fork Salmon			Lemhi River		
	Female	Male	Count	Female	Male	Count	Female	Male	Count	Female	Male	Count
1970	0.436	0.564	133	0.417	0.583	24	0.412	0.588	665	0.483	0.517	87
1972	0.306	0.692	535	0.418	0.582	91	0.504	0.496	262	0.634	0.366	161
1973	0.584	0.416	306	0.541	0.459	37	0.563	0.437	604	0.636	0.364	129
1974	0.494	0.506	164	0.250	0.750	4	0.513	0.487	156	0.788	0.212	33
1975	0.784	0.216	37	1.000	0.000	1	0.500	0.500	152	0.724	0.276	29
1976	0.515	0.485	33	0.000	1.000	5	0.333	0.667	24	0.714	0.286	14
1977	0.044	0.956	250	0.054	0.946	37	0.183	0.817	366	0.500	0.500	16
1978	0.668	0.332	211	0.703	0.297	37	0.876	0.124	404	0.649	0.351	74
1979	0.300	0.700	10	—	—	—	0.563	0.436	16	1.000	0.000	4
1980	0.333	0.667	42	0.250	0.750	4	0.429	0.571	7	—	—	—
1981	0.667	0.333	12	0.571	0.429	14	0.333	0.667	6	0.857	0.143	7
1970-1981 (inverse of of female frequency)	0.407	0.593	1735	0.421	0.579	254	0.471	0.529	2662	0.630	0.370	554
	2.454			2.374			2.125			1.587		
Excluding 1977 (inverse of of female frequency)	0.469	0.531	1485	0.484	0.516	217	0.521	0.479	2296	0.634	0.366	538
	2.134			2.067			1.918			1.576		
1985				0.423	0.577	26	0.257	0.743	113			
1986				0.481	0.519	27	0.489	0.511	64			
1987				0.351	0.649	74	0.366	0.632	117			
1988				0.588	0.412	119	0.550	0.450	433			
All years (inverse of of female frequency)				0.466	0.532	246	0.466	0.532	727			
				2.050			2.136					

Source: IDFG Spawning Ground Surveys



Table 5-16. Continued

SPRING CHINOOK INDEX AREA												
Year	Salmon River			East Fork			Upper Valley Creek			Marsh Creek		
	Female	Male	Count	Female	Male	Count	Female	Male	Count	Female	Male	Count
1970	0.430	0.561	132	0.478	0.522	228	0.400	0.600	15	0.442	0.558	120
1972	0.528	0.472	280	0.530	0.470	151	0.600	0.400	65	0.681	0.319	47
1973	0.567	0.433	252	0.641	0.359	398	0.764	0.236	80	0.633	0.367	120
1974	0.727	0.273	77	0.661	0.439	57	0.605	0.395	43	0.667	0.333	39
1975	0.703	0.297	480	0.531	0.469	81	0.764	0.236	80	0.663	0.337	98
1976	0.515	0.485	97	0.750	0.250	4	---	---	---	0.200	0.800	5
1977	0.327	0.673	743	0.642	0.358	81	---	---	---	0.560	0.440	25
1978	0.703	0.297	760	0.618	0.382	432	0.764	0.206	34	0.785	0.215	205
ten	0.625	0.375	56	0.333	0.667	6	1.000	0.000	1	0.750	0.250	4
1980	---	---	---	---	---	---	---	---	---	---	---	---
1981	0.596	0.404	52	0.750	0.250	8	---	---	---	0.800	0.200	5
1970-1981 (inverse of of female frequency)	0.559	0.441	2927	0.587	0.413	1446	0.699	0.301	336	0.651	0.349	668
1.789				1.703			1.430			1.536		
Excluding 1977 (inverse of of female frequency)	0.636	0.362	2184	0.584	0.416	1365	0.699	0.301	336	0.655	0.345	643
1.568				1.713			1.430			1.527		
1985												
1986												
1987												
1988												
All years (inverse of of female frequency)												

Table 5-16. Continued

SPRING CHINOOK INDEX AREA												
YOU	Elk Creek			Bear Valley			Sulphur Creek			Upper Big Creek		
	Female	Male	Count	Female	Male	Count	Female	Male	Count	Female	Male	Count
1970	0.645	0.355	200	0.535	0.465	286	0.529	0.471	34	0.447	0.553	36
1972	0.484	0.506	170	0.505	0.495	97	0.471	0.529	17	0.846	0.154	13
1973	0.762	0.238	450	0.662	0.338	234	0.631	0.369	103	—	—	—
1974	0.594	0.406	32	0.570	0.430	172	0.633	0.167	6	0.407	0.593	27
1975	0.579	0.421	19	0.633	0.367	30	0.667	0.333	3	0.529	0.471	17
1976	0.286	0.714	7	0.500	0.500	8	0.333	0.667	3	—	—	—
1977	0.216	0.784	74	0.263	0.737	116	0.000	1.000	2	0.143	0.857	7
1978	0.672	0.328	131	0.624	0.376	237	0.633	0.167	6	0.632	0.368	36
1979	0.700	0.300	10	0.818	0.182	11	1.000	0.000	2	1.000	0.000	6
1980	—	—	—	0.583	0.417	12	—	—	—	1.000	0.000	2
1981	0.714	0.286	7	0.875	0.125	8	0.000	1.000	1	—	—	—
1970-1981 (Inverse of of female frequency)	0.640	0.360	1100	0.561	0.439	1213	0.599	0.401	177	0.547	0.453	146
	1.563			1.784			1.670			1.827		
Excluding 1977 (Inverse of of female frequency)	0.671	0.329	1026	0.593	0.407	1095	0.606	0.394	175	0.567	0.433	141
	1.491			1.687			1.651			1.763		
Elk, Bear Valley, Sulphur, Upper Big Creeks Combined												
1985										0.250	0.750	4
1986										0.500	0.410	156
1987										0.555	0.445	110
1988										0.665	0.335	310
All years (Inverse of of female frequency)										0.621	0.379	560
										1.611		

The expansion would be biased if there is a bias in the relative counts of male and female spawners. It would be biased if the relative proportions of carcasses were not equal to those of spawners. Since counts are based on total dead fish, not spawned-out fish; it could be quite biased. And under certain situations, the estimate would not be a good indicator of spawner number. In Table 5-17, we give expansion values used as input into the System Planning Model along with our estimate from 1977, from the 1970-1981 data set excluding the 1977 data, and from the 1985-1988 data set.

Table 5-17. Spawner expansion factors for redds based on different data sets		
Area/Population	Source	Expansion Factor
South Fork Salmon, Johnson Creek (summer chinook)	SSPM	2.31
	1977	9.46
	1970-81 excluding 1977	2.00
	1985-1988	2.11
Middle Fork Salmon (Bear Valley, Elk, Sulfur, and Upper Big creeks): spring chinook	SSPM	1.82
	1977	4.18
	1970-81 excluding 1977	1.60
	1977	1.61

The year 1977 was a very unusual year in that the number of males was unusually high, especially for summer chinook for which the male frequency ranged from 85% to 96% over the three summer chinook index areas. The system is dynamic enough that for the purpose of assessing recovery program, estimates needed to generate brood year contributions should be, to the extent possible, real-time estimates not long-term estimates or model estimates. However, biases can be generated by abnormal distributions. It is highly unlikely that multiplying the number of redds by 9.46 in the case of summer chinook or by 4.18 in the case of spring chinook gives any indication as to the number of effective spawners. We note that the estimates from 1979-1981 excluding 1977 and from 1985-1988 are very similar, both for the spring and the summer chinook, and they are lower than that

used as input to System Planning Model. It may be that the input estimate was based on data that included the 1977 data set. The 1977 data set not only included an unusually high male frequency, but it also included relatively high returns. This could have resulted in an estimate that is higher than would apply to most years.

**RECOMMENDATION: In assessing the recovery program, we recommend that redd counts not be expanded by spawner number, and that log ratios between unexpended return and brood-year redds be used to assess trend.**

### **5.2.2 Spawner trend**

In order to examine the net effects of changes in survival rates downstream from the spawning and rearing areas (that is to exclude the effects of habitat degradation within the natal stream) we examined the trends in indices of spawner escapement in Snake River tributaries that lie in wilderness areas. For example, counts of spring chinook redds in the Wenaha River and the Minam River (both wilderness areas in northeast Oregon) show a sharp decrease in abundance during 1960 to 1980 (Figure 5-11), despite the substantial drop in harvest rates in the Columbia River. Similarly, the index of redds per mile averaged over 14 streams in the Salmon River Basin that are managed for wild fish has dropped steadily over the same period (Figure 5-12). We used a Poisson regression to estimate the average rate of decline in these wild populations, and found that the decline rate was highly significant ( $P < 0.01$ ) on each stream and ranged from 8% to 11% per year (Figure 5-13). We did not include data after 1983 in these trend analyses, because hatchery fish began straying into the wilderness areas in the mid 1980's and inflated the redd counts.

We calculated a Poisson regression of the spawning escapement predicted from our simulation on the years from 1964 to 1983 and found the decline rate was 13.7% per year, slightly higher than that observed in wilderness areas (Figure 5-13). Thus, the population model we used should be regarded as liberal in its allowances for the effects of dams and downstream harvest. Our simulation model, which was developed to evaluate rebuilding schedules is detailed in a later section of this report.

## Spring Chinook Spawner Trends Wilderness Areas

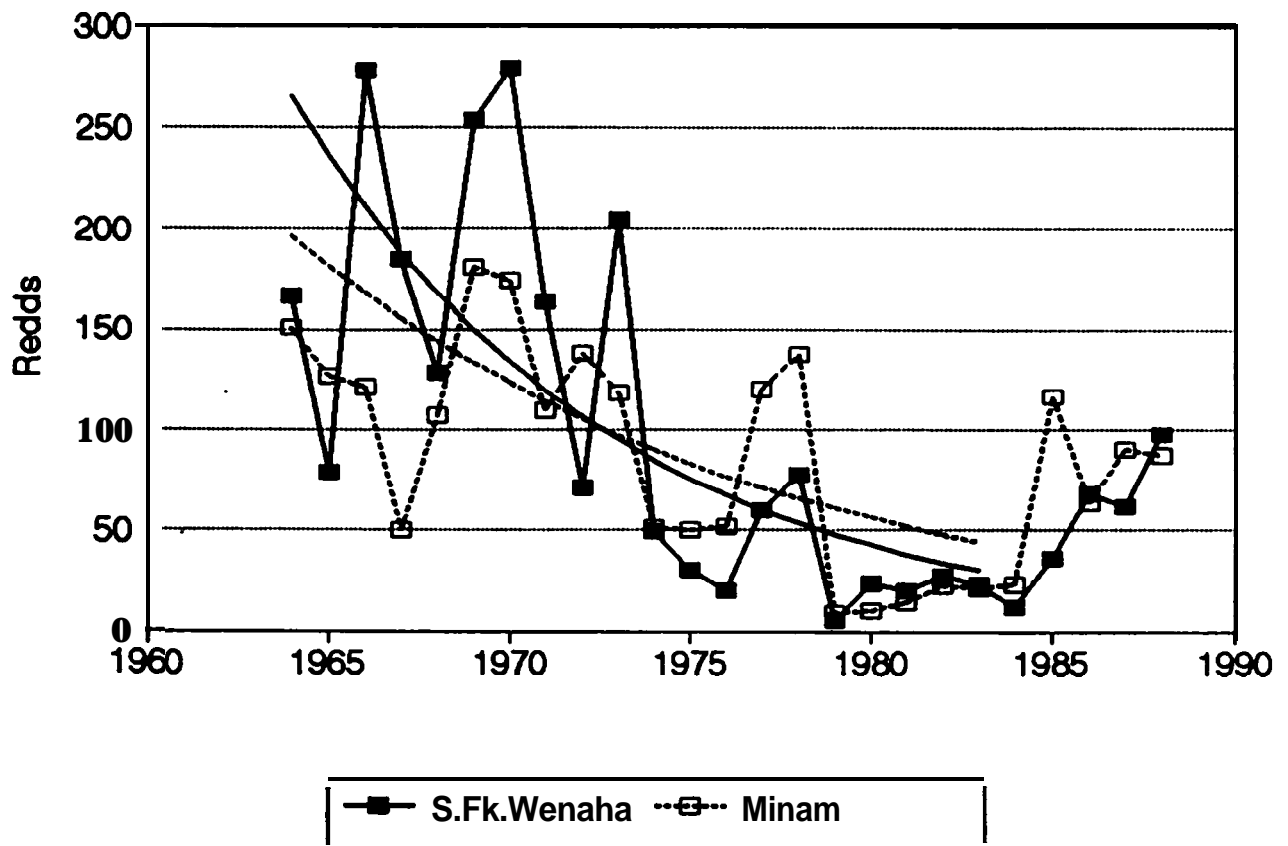


Figure 5-11. Observed trend in counts of spring chinook spawning redds in the South Fork Wenaha and Minam Rivers during 1964-1983. Both streams are in wilderness areas. Data from Grande Ronde Subbasin Plan (ODFW 1989). Poisson regression equation was  $\text{Redds} = 230.5 * e^{-.1145 * \text{Year}}$  for the Wenaha and  $\text{Redds} = 159.3 * e^{-.0785 * \text{Year}}$  for the Minam.

## Spring Chinook Spawner Trends

### Salmon River Trend Streams

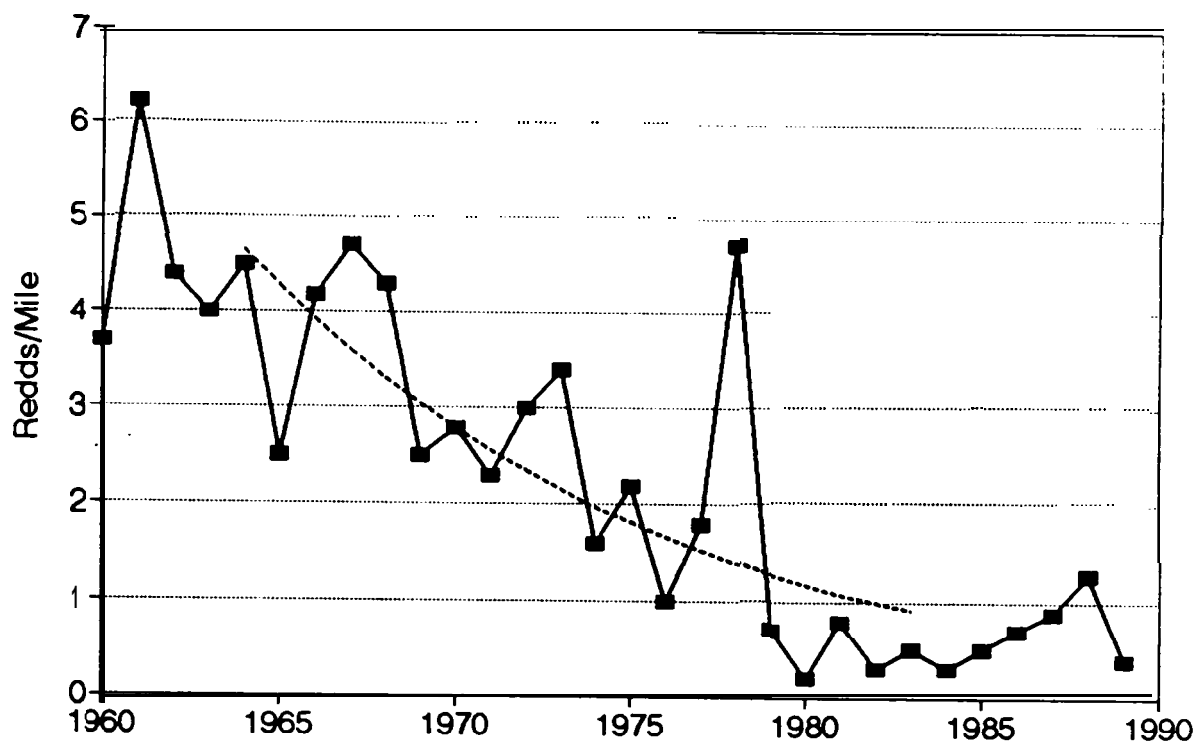
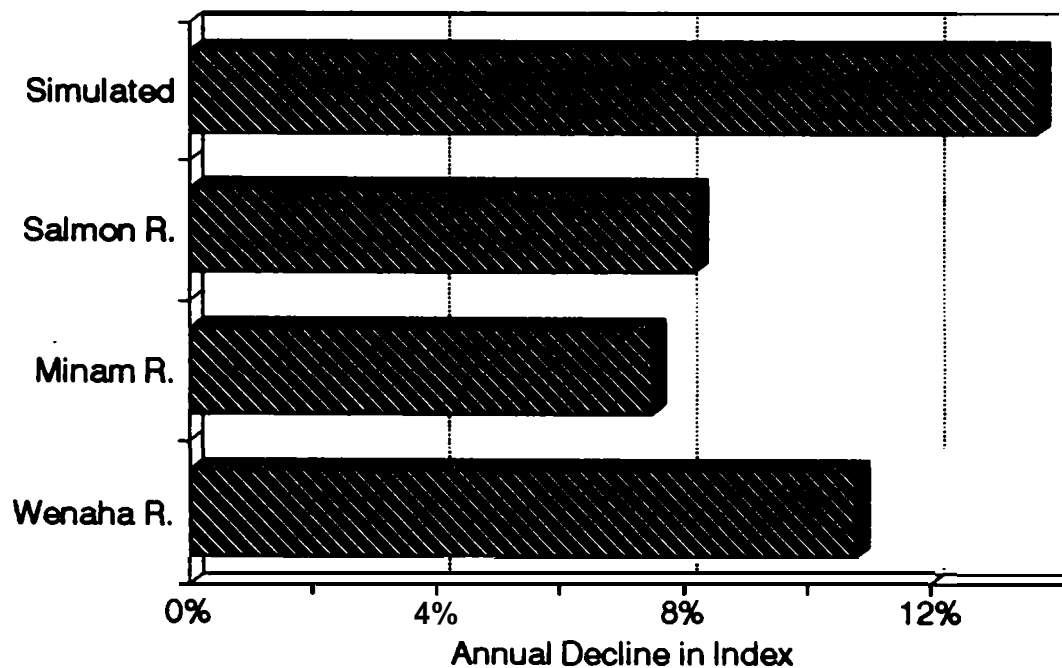


Figure 5-12. Observed trend in the counts of spring chinook spawning redds averaged for 14 streams in the Salmon River Basin that are managed for wild/natural fish. Data from ODFW and WDF 1991. Poisson regression equation was  $\text{Redds/Mile} = 169.5 * e^{-.0855 * \text{Year}}$

## Spring Chinook Spawner Trends

Average Decline Rates, 1964-1983



**Figure 5-13.** Comparison of rates of decline in spawning redds of spring chinook in wild index areas with those simulated for the Snake River. Annual rates of decline estimated by Poisson regression for 1964-1983 data.

### 5.3 MEASURES OF SURVIVAL

#### 5.3.1 Parr-to-smolt

Estimates of parr-to-smolt survival are imprecise, difficult to obtain, and have been rarely attempted. We do not recommend use of this survival rate as a de-listing criteria. For the purposes of our rebuilding analysis, we reviewed estimates that have been made in the Columbia Basin and concluded that overwinter survival for chinook parr averages about 30%. Some workers have estimated lower survival rates, and we suspect that many such estimates are biased low by a lack of accounting for parr that emigrate from the study area in the fall and winter. We are particularly skeptical of survival estimates to Lower Granite Dam (they are consistently less than appears reasonable) and we suspect that the fraction of fish sampled there is underestimated. A synopsis of the information that lead us to estimate parr-to-smolt survival at 30% follows.

- ▶ Petrosky (1990) used PIT tag data in the upper Salmon River to estimate 26% overwinter (parr-to-smolt) survival for parr that remained in the study area over winter. He also found that 67% of the parr migrated out of the study area and past the Sawtooth Hatchery weir in the fall.
- ▶ Petrosky (1990) used PIT tags on chinook parr in Crooked River (tributary of the South Fork Clearwater) and estimated that overwinter survival was 31%. Only 16% of parr left Crooked River in the fall.
- ▶ Fast et al. (1991) found overwinter survival for juvenile spring chinook of 22% to 49% annually during five winters for Naches winter migrants to Prosser in the Yakima River. Survival of migrants branded at Rosa in the winter was 44.9% in 1989-90.
- ▶ Lindsay et al. (1989) estimated 52% overwinter survival in the mainstem Deschutes River (not in the Warm Springs River) based on CWTs of wild fish emigrating from the Warm Springs River in fall and spring, 1978 brood. However, Lindsay et al. qualify this estimate, because fall migrants that overwintered in the Deschutes River were bigger in spring than those in the Warm Springs River, and their 52% estimate assumed equal survival from **smolt** to adult of both groups. Lindsay et al. (1989) **estimated that overwinter survival was 20% for fish remaining in the Warm Springs River.**



- 7 Lindsay et al. (1985) estimated that overwinter survival of spring chinook parr averaged 30% in the John Day River during the 3 yr of study.

### 5.3.2 **Downstream Passage**

Estimates of juvenile losses during passage through the Snake and Columbia Rivers vary widely from year to year and are affected by numerous factors, many of which change at each dam and reservoir. The possible ways that bias might enter estimates of loss are legion. Yet, it is clear that increases in downstream mortality since pre-dam times have been a dominant cause of the population decline. Therefore, it is unavoidable that this migration mortality factor be seriously considered as a de-listing criteria. Because of the numerous variables that influence downstream survival, we suggest that an estimate of overall survival from the time smolts leave their natal stream until they have passed all dams is the most useful survival measure as a possible de-listing criterion.

For the purposes of our rebuilding analysis, we chose to use a loss rate per project (dam plus impoundment) of 15%, because it is the most widely accepted per project mortality rate for yearling chinook, including direct and indirect causes of mortality (NPPC 1989). Raymond (1988), who has studied survival rates of smolts in the Snake and Columbia Rivers perhaps more than any other worker, used 15% loss per project to adjust his smolt-to-adult survival data between years in which there were different numbers of dams in the migratory pathway. Raymond (1969) estimated that during 1966 through 1968, mortalities of yearling chinook migrating downstream from the Salmon River to the Dalles Dam averaged 40% (based on comparable marked groups released in the Ice Harbor Dam forebay and John Day Dam tailrace). This average represents about 15% loss per project, given that these fish passed through three reservoirs and over two dams. Similarly, the CRiSP model was recently used to estimate that the average survival rate of all outmigrating spring chinook smolts under 1990 migration conditions was 18.8% past eight dams (Fisher 1992). A 15% loss per project would reduce survival past eight projects to 27% due to dam related factors only, so the two estimates appear congruent.

This loss rate per dam was not intended to account for all mortality to downstream migrants, because some mortality certainly occurred before dams were built. Rather, the **average loss rate per dam** was used as a scaling factor in our simulation model to account for the reduction in survival that **occurred** with each new dam. **The** reduction in survival was subtracted from the smolt-to-adult survival rate, which already included the natural mortality that occurred during downstream migration during pre-dam times.

### **5.3.3 Smolt-to-Adult Survival Rates**

The smolt-to-adult survival rates of wild Snake River spring chinook was estimated each year of smolt outmigration from 1964 to 1984 by Raymond (1988). Raymond relied on extensive mark-recapture sampling coordinated by the National Marine Fisheries Service at mainstem dams and on hatchery release and return data to develop his estimates. We accept Raymond's estimates (Figure 5-14) as reasonable and the best available. We used Raymond's estimates of smolt-to-adult survival rate during the time when only four mainstem dams were in place (1964-1968) to back calculate the survival rates that would have occurred before dams were built. The average of Raymond's survival estimates for the 1964-1968 smolt outmigrations was 4.2%. These estimates were based on the number of smolts at Ice Harbor Dam in the Snake River and the number of adults returning to Ice Harbor Dam, including river harvest. Raymond assumed that ocean harvest was negligible, and recent analyses of CWT recoveries from Snake River spring/summer chinook confirm that the harvest rate of these fish in the ocean is less than 1% (Berkson 1991).

In order to convert Raymond's (1988) smolt-to-adult values to the equivalent of pre-dam times, we had to adjust them for juvenile and adult losses at each dam. The smolts he was working with passed four dams and nearly all of the returning adults had to pass five dams, before they were at the point at which he estimated their abundance. Adults returned primarily after 3 yr in the ocean (3-salt), which means they returned upriver during 1967-1971. John Day Dam was completed in 1968, so adult numbers at Ice Harbor reflected fish that had passed four Columbia Dams plus Ice Harbor. Therefore, if we assume a 15% loss of smolts per dam and a 5% loss of adults per dam, the pre-dam smolt-to-adult survival rate becomes  $4.2\% / (.85^4) \times (.95^5) = 10.4\%$ .

## Smolt-to-Adult Return Rates, Snake R. Wild Spring Chinook - Raymond (1988)

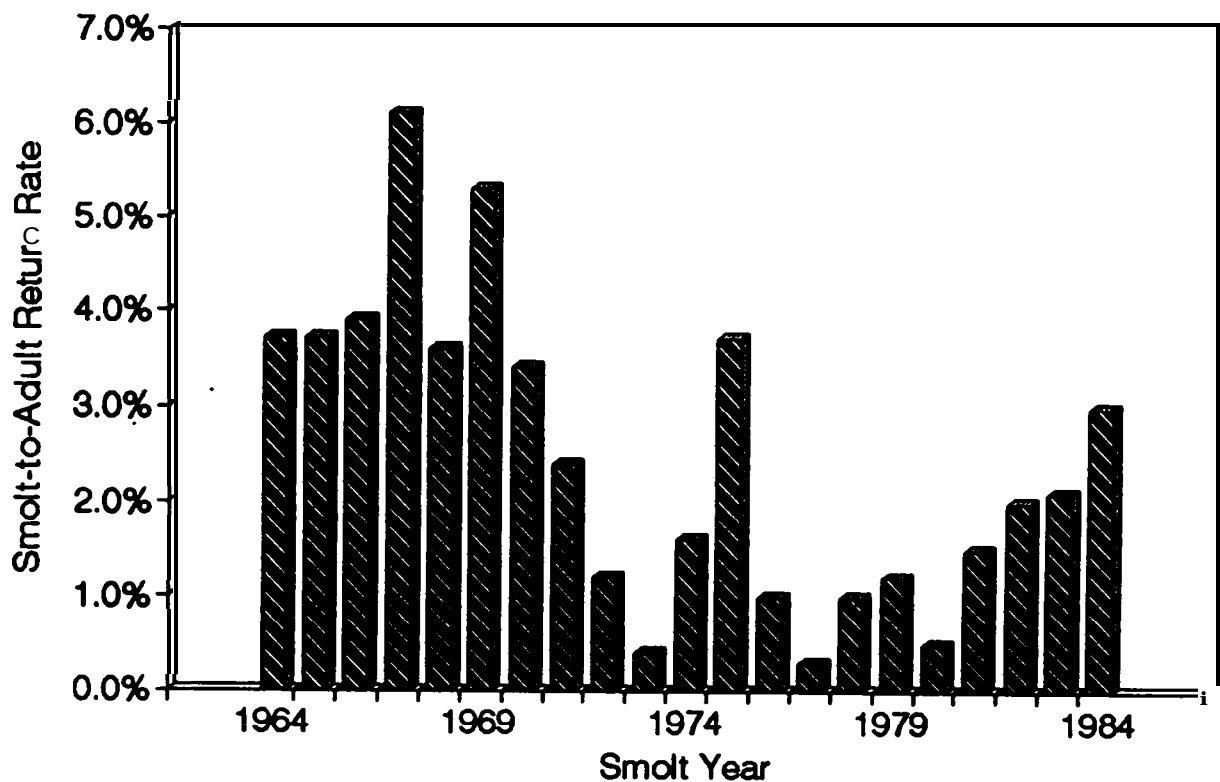


Figure 5-14. Smolt-to-adult survival rates for wild yearling chinook (hatchery fish excluded) passing Ice Harbor Dam, as reported by Raymond (1988).

We checked our 10.4% survival estimate against estimates for several other spring chinook populations in the Columbia Basin and found there was a high level of agreement. These estimates with which we found agreement include Petrosky (1991), Lindsay et al. (1989), Fast et al. (1991). Petrosky (1991) estimated smolt-to-adult survival rates for spring chinook from Marsh Creek (Salmon River tributary) for the 1958-85 broods. Petrosky had a long time series of redd counts in Marsh Creek, but did not have any data on juvenile abundance, so he predicted the abundance of parr each year by applying the spawner abundance to the egg-to-parr function from the System Planning Model. For his input to the egg-to-parr function, he estimates a carrying capacity equivalent to 67 fish/100m<sup>2</sup>, which indicates that he must have been estimating parr rather than smolts. Petrosky assumed total upstream survival was reduced by 10% per dam in the Columbia River and 5% per dam in the Snake River. Petrosky also included an adjustment for in-river harvest.

Smolt-to-adult survival rates estimated by Petrosky (1991) and Raymond (1988) are highly correlated ( $r = 0.93$ ), except for the three broods in which Petrosky estimated survivals greater than 3% (Figure 5-15). In two of those brood years, 1962 and 1968, the data used by Petrosky may have been subject to large sampling error with regard to age distribution. In those years the samples indicated the highest proportion of 2-salt spawners (52% and 50%) of any of the 28 broods sampled. Returns from the third unusual brood, 1980, were estimated primarily from the record low count of only 9 redds in Marsh Creek, so if surveyors had missed a few redds, perhaps by not counting at the peak of spawning, Petrosky would have substantially underestimated the number of smolts, thereby overestimating smolt-to-adult survival rate. Data for all years were subject to sampling error, so we cannot be sure that these explanations of the flyer points in Figure 5-15 are accurate, but they seem reasonable.

## Smolt-to-Adult Return Rates, Snake R.

Raymond (1988) v. Petrosky (1991)

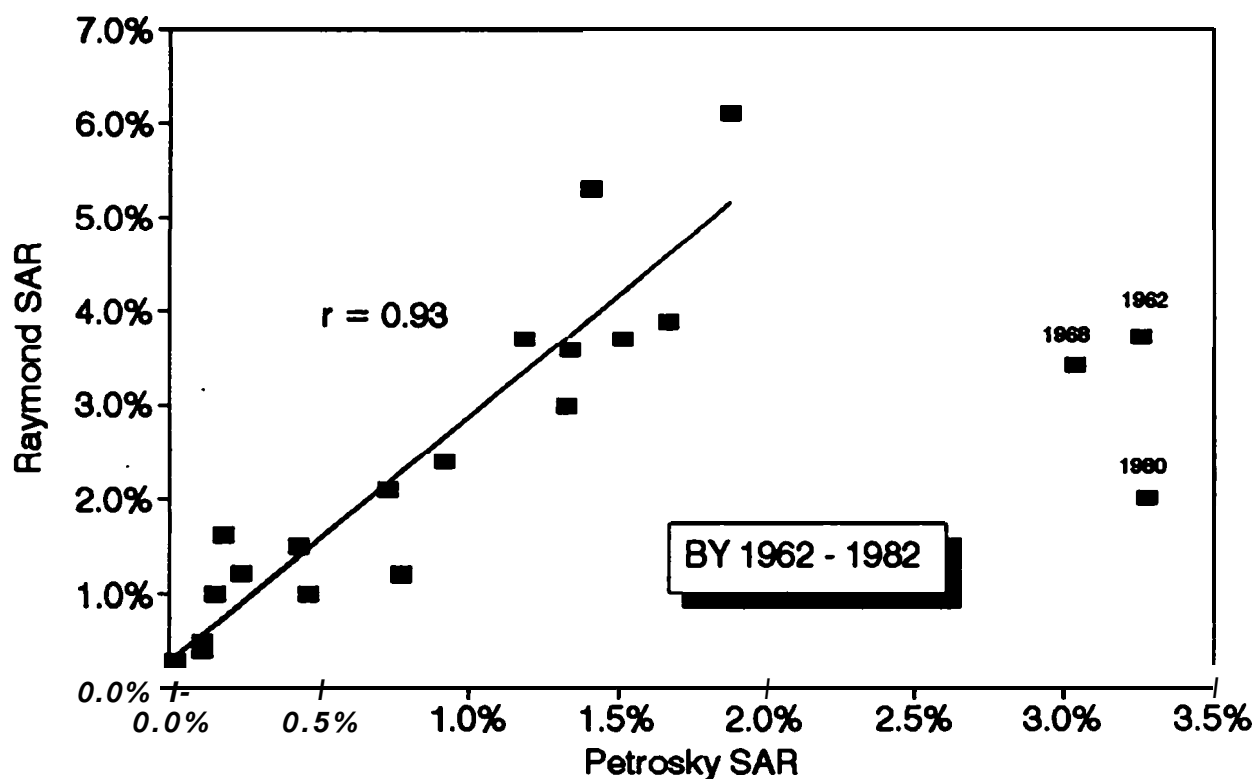


Figure 5-15. Correlation between estimates by Raymond (1988) and Petrosky (1991) of smolt-to-adult survival rates for Snake River spring chinook. Labeled points show brood year.

The high correlation between all but three of the estimates by Raymond and Petrosky lends validation to both methods, but the slope of the regression indicates that Raymond's estimates are about 2.6 times higher than Petrosky's. We found that this difference did not reflect different estimates of survival, but rather reflected inequivalency of the measures used by the two workers for smolts and adults. Petrosky used substantially larger smolt units than Raymond, because Petrosky used an estimate of parr in the fall and Raymond used smolts in the spring at the uppermost Snake River dam. We estimate that only about 30% of the parr estimated by Petrosky survived to smolting. Petrosky's parr would also have suffered another 11% mortality to the first dam (Raymond 1988), where Raymond began his estimates of smolts. On the other hand, Raymond used lower expansion factors to estimate adults returning to the Columbia River. Raymond expanded his estimates for lower river catch, but not for upstream loss at each dam. Petrosky applied a 10% mortality for adult loss at Columbia River dams and 5% loss at Snake River Dams. Petrosky used actual spawners for his estimates of escapement, while Raymond used adults returning to Ice Harbor Dam. The net effect of adjusting the estimates by Petrosky and Raymond to equivalent terms would be to multiply Petrosky's estimates by 2.6, the slope of the regression line between the two.

Lindsay et al. (1989) estimated that smolt-to-adult survival rates of wild spring chinook in the Warm Springs River averaged 3% during 1975-1981. This estimate included fall and spring migrants with no weighting of the two groups. We adjusted Lindsay et al.'s numbers according to their own estimates of 30% overwinter survival of fall migrants and 10% ocean harvest plus 15% Columbia River harvest. Adjusted smolt-to-adult survival rates for 7 years (1975-81 broods) averaged 10% and ranged from 5 to 15%. This is highly consistent with our pre-dam estimates based on the data of Raymond (1988).

Fast et al. (1991) estimated that smolt-to-adult survival of wild spring chinook in the Yakima River averaged 3.75% over five broods (1981-1985). These rates did not include accounting for losses of adults at dams, in the ocean harvest or in the Columbia River harvest. Their CWT data showed 25.7% harvest in Columbia and 1.6% harvest in ocean. Once we adjusted the recruits for these factors their survivals to the mouth of the Columbia ranged from 5.7% to 19.5% and averaged 12.2% for the five broods. Again, this is highly consistent with our pre-dam estimates based on the data of Raymond (1988).

#### **5.3.4 Harvest rates**

We used empirical estimates of harvest rate in our simulation modeling for all years possible. Analyses of CWT recoveries from Snake River spring chinook indicated that harvest rate in ocean was less than 1% (Berkson 1991). Therefore, we ignored ocean harvest. We obtained estimates of in-river harvest rates each year from ODFW and WDF (1991). We assumed that spring chinook from the Snake River were harvested at the same rate as the upriver spring chinook runs for which harvest and escapement have been summarized by ODFW and WDF (Figure 5-16). Harvest rate data were not available prior to 1938, so we assumed harvest rates were equal to the average for the period, 1939-1956, when harvest rates were fairly stable. The mean harvest rates for these years were 46% in zones 1-5 and 25.8% in zone 6. The assumption that these rates can be hind-casted appears reasonable, based on the work of Johnson et al. (1948) who studied the effects of changes that had been made in harvest regulations since the early 1920's and concluded, "that the elimination of any one type of gear on the Columbia River has served only to increase the catch by other gears rather than increase the escapement." Sport harvest in the Snake River and tributaries has been negligible since 1975, but should be added to our simulations prior to 1975 (Horner and Bjorn 1981).

#### **5.3.5 Upstream Passage**

We used a mortality rate of 5% per dam for simulations of adults migrating upstream. Chapman et al. (1991) reviewed estimates of inter-dam loss and concluded 95% survival per dam was the most reasonable estimate. NMFS (1992) estimated that survival of spring chinook past all eight dams into the Snake River Basin was 66%, which is equivalent to 95%/dam.

The net effect of upstream and downstream losses increased steadily from 1938 to 1980 (Figure 5-17). By the late 1960's, mortality had increased sufficiently that restrictions were added to harvest. By the late 1970's, nearly all harvest had ceased and the estimated losses during juvenile and adult migrations often exceeded 90%. The combined effect of harvest and in-river losses during juvenile and adult migrations caused a sharp decline in the survival of spring chinook, relative to their survival under pristine conditions (no impoundments and no harvest) (Figure 5-18).

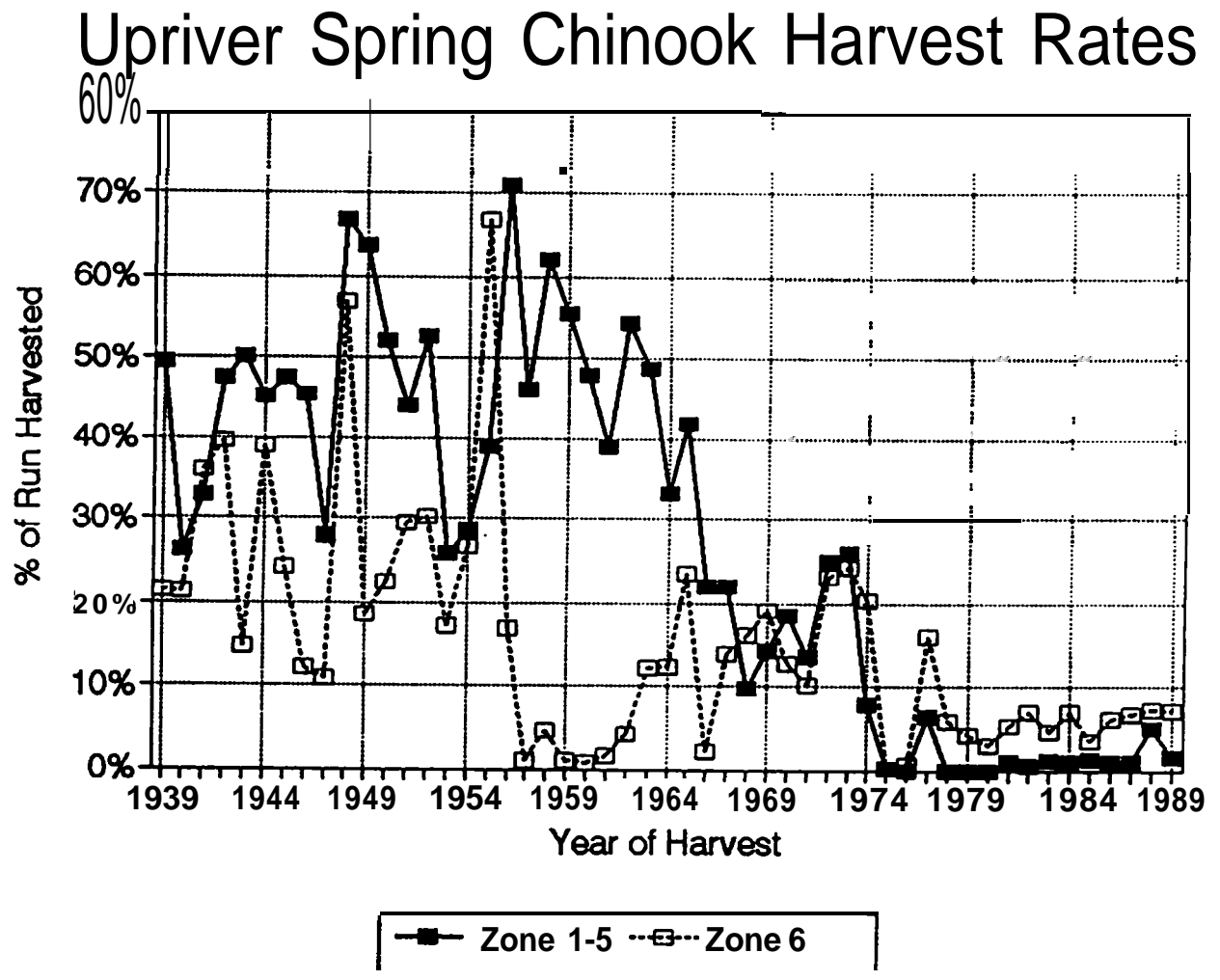


Figure 5-16. Harvest rates on upriver spring chinook in Columbia River commercial fisheries, 1938-89 (data from ODFW and WDF 1991).



## Survival Trends for Spring Chinook

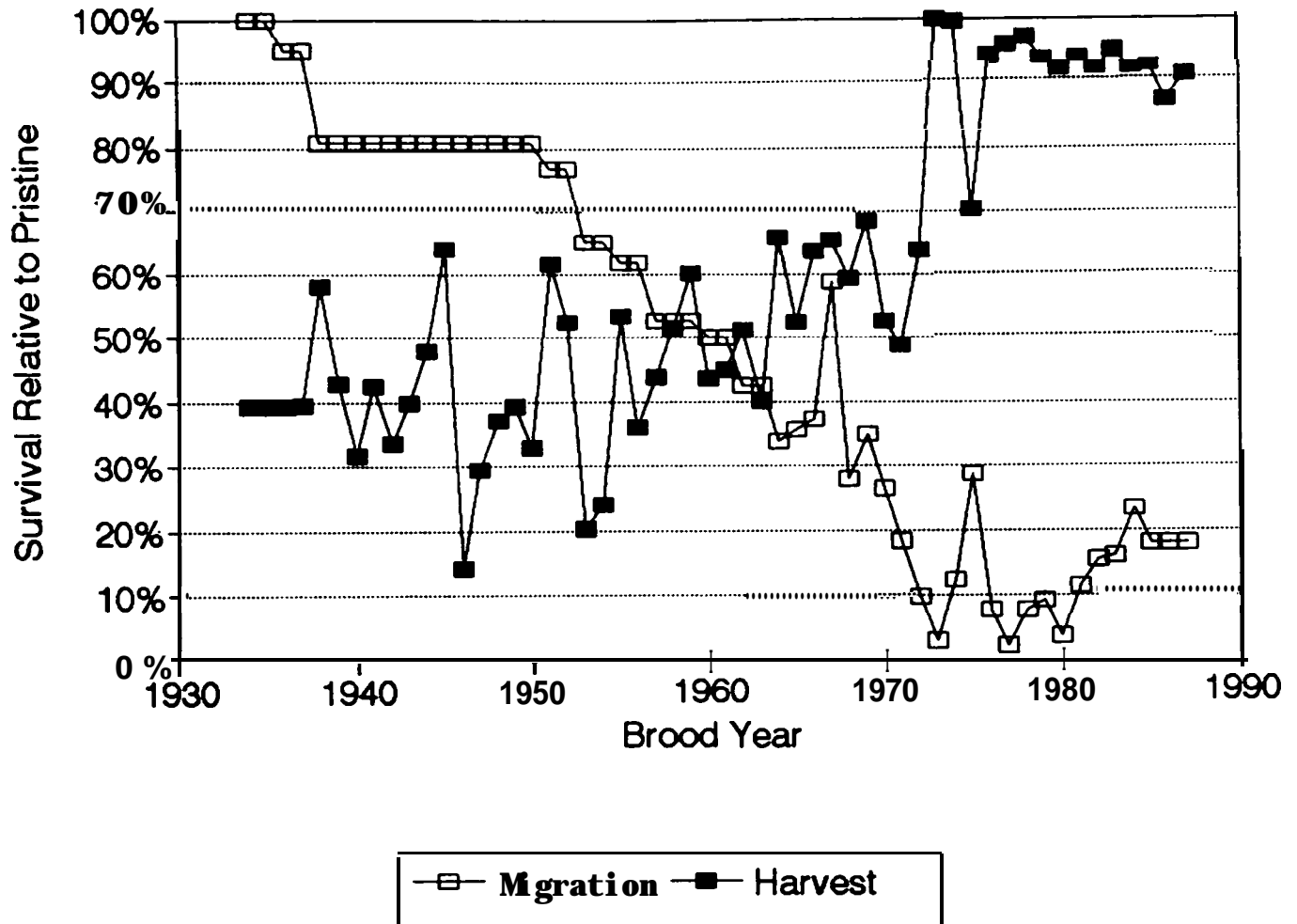
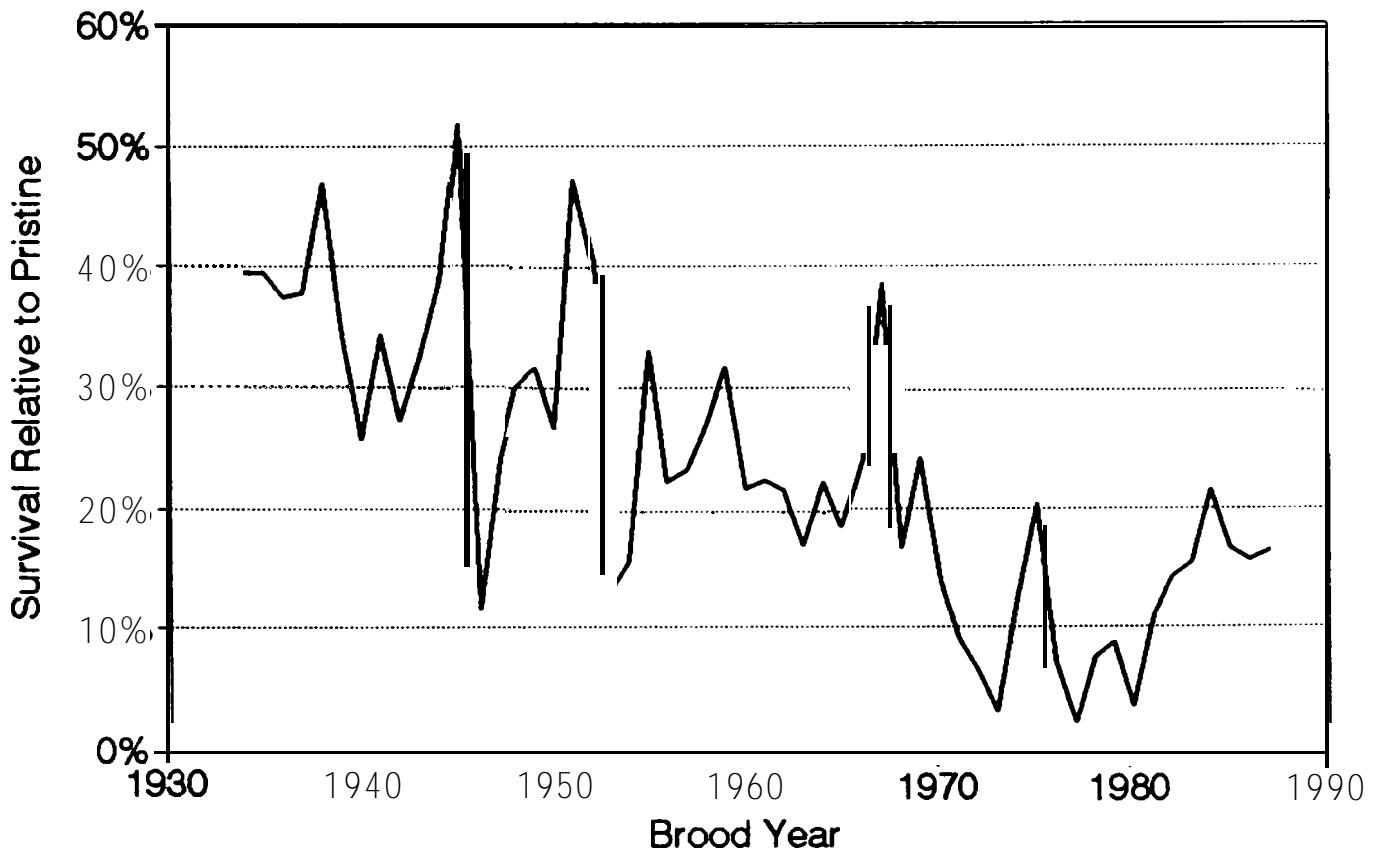


Figure 5-17. Annual changes in mortality of Snake River spring chinook that were estimated to result from harvest and in-river losses during juvenile and adult migrations, 1928-92.

## Survival Trends for Spring Chinook Harvest and Migration Losses Combined



**Figure 5-18.** Trend in estimated survival of spring chinook during 1928-1992. Survival is expressed as a percentage of the survival under pre-dam conditions with no harvest, and includes the effects of harvest and in-river losses during juvenile and adult migrations.

## **6. DE-LISTING CRITERIA FOR FALL CHINOOK**

### **6.1 STOCK-RECRUITMENT RELATIONSHIPS AND CARRYING CAPACITY**

#### **6.1.1 Review of Literature Values of $\alpha$**

There appears to be no data from which to directly estimate the Ricker  $\alpha$  parameter for fall chinook in the Snake River Basin, so we evaluated the usefulness of parameters that had been estimated from data on other fall chinook populations. Chapman et al. (1982) estimated that the Ricker  $\alpha$  parameter for the population of fall chinook in the Columbia Basin above Bonneville Dam was 26.1 for the 1938-1946 broods. These broods preceded the construction of all mainstem dams that fish could pass, except Bonneville Dam and Rock Island Dam. Healey (1982) estimated Ricker function parameters for all chinook stocks in British Columbia combined for the 1951-1976 broods, based on ocean harvest data for the entire province. Healey reported that over 90% of the catch was fish that had entered the ocean as subyearlings, which indicates that they were fall chinook. Healey's estimates of the Ricker  $\alpha$  parameter ranged from 11.3 to 20.1. Schaller and Cooney (1992) used data on fall chinook from the mid-Columbia to estimate the Ricker  $\alpha$  parameter for Snake River fall chinook under present day conditions; their estimate was 7.2.

#### **6.1.2 Adjustment of Ricker $\alpha$ Parameter for Equivalency**

Although these estimates of the Ricker  $\alpha$  parameter for fall chinook appear to be widely divergent, they actually are quite similar, once the data on Parents and Recruits are adjusted to pre-dam conditions. It was important to our purpose to estimate the pre-dam stock-recruitment function, so that in our hindcasting of recruitment, we could add the incremental effect of each dam as it was built. We adjusted Parent values to the equivalent of successful spawners, and the Recruit values to the equivalent of adults which would have been caught in the ocean or returned to the mouth of the Columbia River before dams were built (Table 6-1). To do this, we made adjustments to account for (1) smolt loss per dam, (2) adult loss per dam, and (3) ocean harvest. Chapman et al. (1982) accounted for all in-river harvest, but not for ocean harvest. Chapman et al. (1982) present data on ocean harvest that indicate about 25% of the adult recruits were caught in the ocean from the 1938-1946 broods, and that this harvest increased to about 55% of adult recruits from the

1947-59 broods. Chapman et al. also made no adjustment for losses of juveniles or adults passing Bonneville Dam. Schaller and Cooney (1992) accounted for all harvest and for adult losses at dams, but not for losses of juveniles at dams. Healey (1982) accounted for all harvest and was dealing predominantly with fish that did not have to pass dams. We used the parameter values from our population model to adjust  $\alpha$  for the unaccounted losses, as just described. The net result was that the adjusted  $\alpha$  from Schaller and Cooney (19.2) was close to the upper range for  $\alpha$  estimated by Healey and to the adjusted value from Chapman et al. for the 1947-59 broods (22.1). However, the adjusted value of Chapman et al. for the 1938-46 broods (39) was substantially higher than other estimates. The cause of the high estimate of  $\alpha$  for the 1938-46 Columbia River broods is unclear, but Chapman et al. concluded that the value for the 1947-59 broods was more likely close to the true value. We conclude from these comparisons that an  $\alpha$  value of 20 is a reasonable estimate for Snake River fall chinook before dams were built. This is equivalent to an unadjusted  $\alpha$  of 7.2 under present-day conditions.

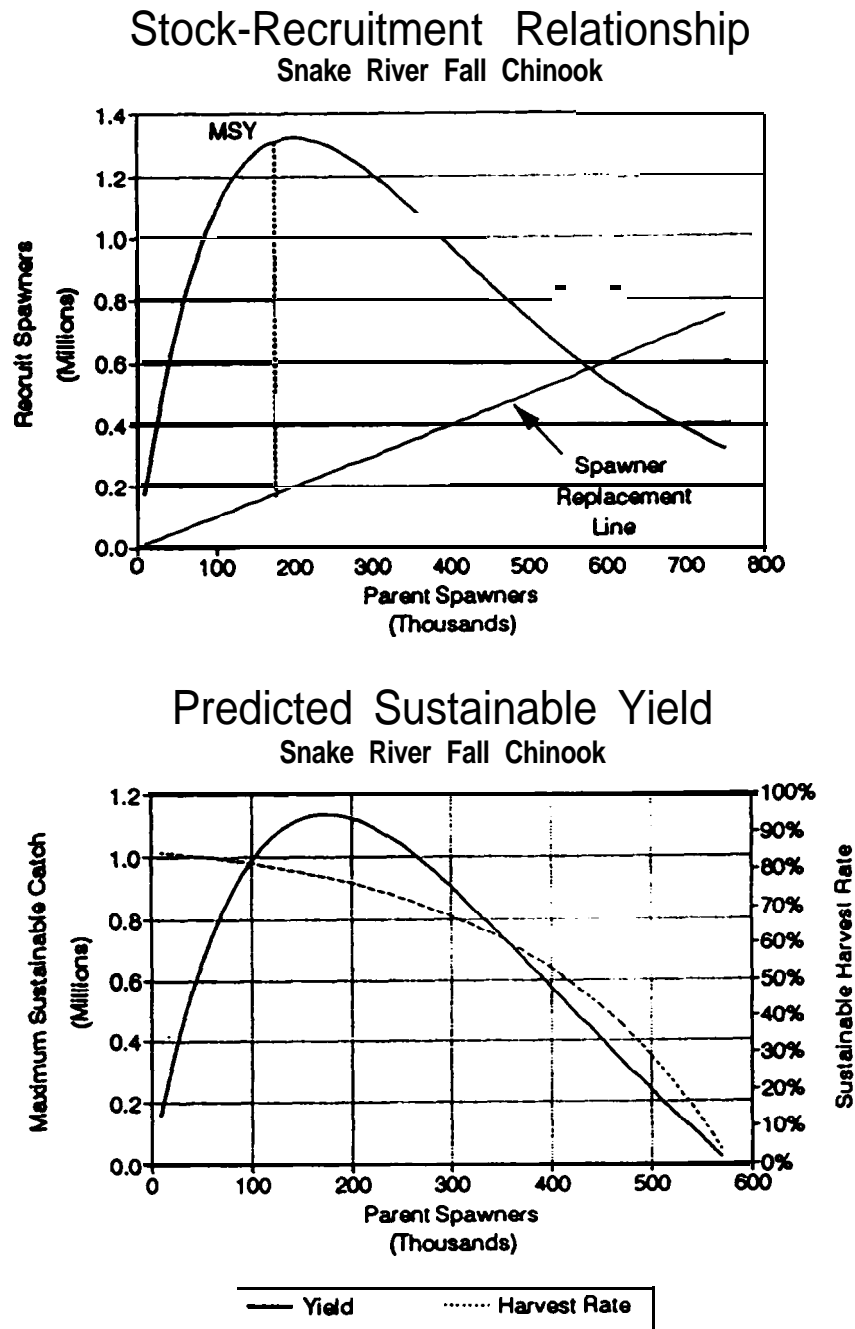
**Table 6-1. Reported values of the Ricker  $\alpha$  value for fall chinook, and adjustment factors that we applied to make their units of measure equivalent. Measures of survival discussed in Section 6.3**

Fall Chinook

Source	Brood Years	Alpha	Juvenile Passage	River Harvest	Adult Passage	Ocean Harvest	Prespawn Survival	Pristine Adjusted Alpha
Healey	51-76	20.00	1.00	1.00	1.00	1.00	1.00	20.00
Chapman et al.	38-46	26.40	0.76	1.00	0.95	0.75	1.00	47.50
chapmanetal.	47-59	7.40	0.76	1.00	0.95	0.45	1.00	22.19
PSC	1980'S	7.10	0.37	1.00	0.81	1.00	1.00	23.55

### 6-1.3 Characteristics of the Stock Recruitment Function for Snake River Fall Chinook

The stock-recruitment function for Snake River fall chinook, based on its Ricker parameter values and the survival rates specified previously, indicates that the pre-dam population would have had a maximum recruitment of about 1.35 million chinook produced by about 200,000 spawners (Figure 6-1). As sources of mortality increased, such as from passage mortality at dams, maximum recruitment and maximum sustainable yield would have declined (see Figure 5-2). Once the cumulative effect of various losses mounts up to 90% mortality or more, the population has little harvestable surplus and will barely maintain itself.



**Figure 6-1.** Stock-recruitment and yield functions estimated for fall chinook in the Snake River Basin under pre-dam conditions.

The stock-recruitment function shows a precipitous drop in surplus production as harvest rate increases beyond that which produces MSY (Figure 6-2). Passage mortality has the same effect as harvest. Once the harvest rate, or the mortality rate over and above the pristine level, mounts up to more than about 80%, the sustainable yield (or surplus production) drops steeply. This indicates that only a few percentage points change in mortality can mean the difference between a population in collapse and a population that is near carrying capacity. Thus, a substantial increase in abundance of spawners should be achieved before we can be confident that the population is safe from extinction. This is discussed further in Chapter 8 on Simulation of Rebuilding.

## Harvest Rate vs Yield

### Snake River Fall Chinook

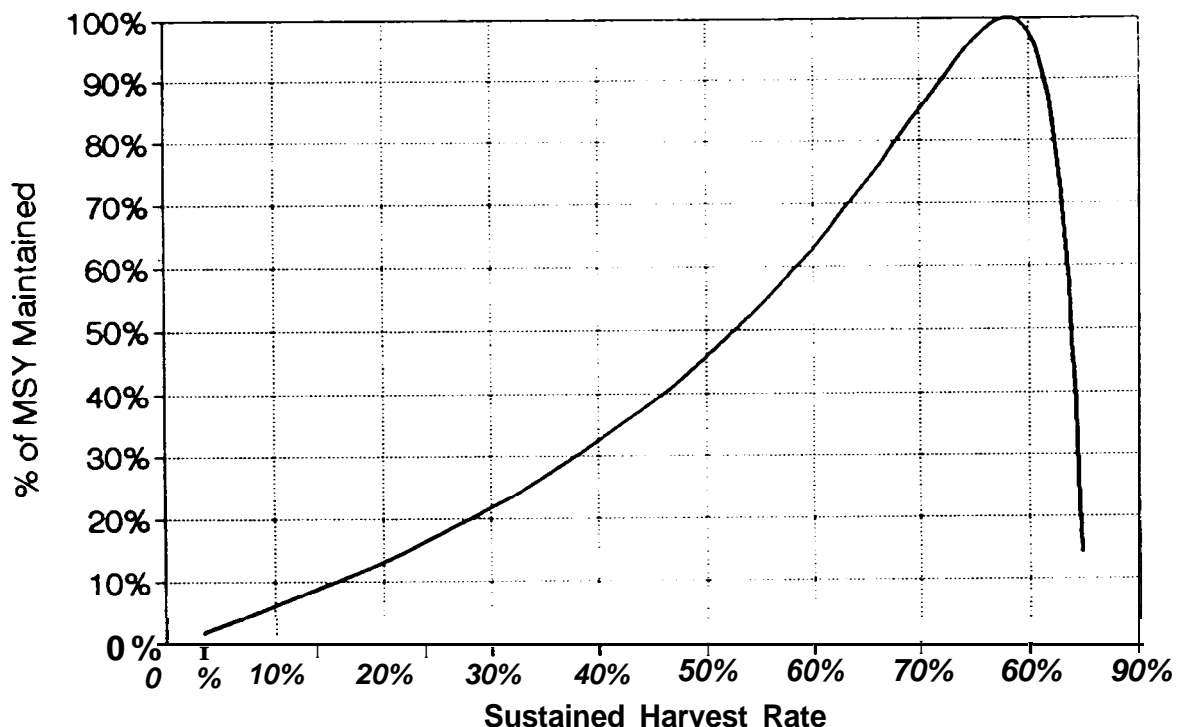


Figure 6-2. Relationship between the proportion of the pristine MSY that can be maintained and the harvest rate plus added mortality. "Pristine" means that passage survival was equivalent to that prior to the advent of hatcheries and mainstem dams.

#### **6.1.4 Carrying Capacity and the Ricker $\beta$ Parameter**

We used our estimate of the carrying capacity of the basin for fall chinook spawners to derive the  $\beta$  value for the Ricker curve. We assumed that production of fall chinook was limited by spawning habitat, such that, the maximum number of recruits that could be produced was determined by the maximum number of spawning pairs the basin could support. This differs from our assumption for spring chinook, for which we assumed the rearing habitat was limiting. Spring chinook juveniles rear in their natal streams through an entire year, while fall chinook parr begin their downstream migration to the sea after only a short period of rearing, perhaps as early as June. Some move directly downstream as emergent fry. Juvenile fall chinook are, therefore, much smaller and require less rearing space in June than spring chinook parr in September. Rearing densities of juvenile fall chinook in late June in the Wenatchee River averaged 40 times greater than that for spring chinook parr in August and September (Hillman and Chapman 1989). Lister and Genoe (1970) found rearing densities of juvenile fall chinook in the Big Qualicum River, British Columbia, as high as 720 fish/m<sup>2</sup>. Juvenile fall chinook spawned in the Snake River Basin may also drift downstream and rear in the impoundments, as has been found for fall chinook in the mid-Columbia. Thus, we conclude that spawning habitat, not rearing habitat, would limit the production of fall chinook in the Snake River Basin.

We found only rough information at best to indicate the carrying capacity of the basin for fall chinook. The majority of fall chinook in the Snake River during the 1950's spawned above Hells Canyon Dam, where their passage is now blocked. Additionally, only anecdotal information exists on the abundance of fall chinook in the Clearwater River prior to 1927, when all chinook populations in the basin were eliminated by the construction of Lewiston Dam. Arnsberg and Connor (1992) estimated that the Clearwater River below the confluence of the North Fork alone could support 191,000 fall chinook spawners, and it is likely that substantial additional habitat suited to fall chinook exists elsewhere in the Clearwater Basin. Schaller and Cooney (1992) estimated that in the mainstem Snake River from Hells Canyon Dam down to Lower Granite Dam, 4,800 fall chinook spawners would produce maximum recruitment.

**RECOMMENDATION: Although the Clearwater Basin has been essentially out of production for fall chinook since 1927, it now represents by far the largest segment of usable habitat for Snake River fall chinook, so we recommend that its habitat capacity be included in the assessment of population recovery (given that the Recovery Plan enables production of fall chinook to expand there).**

After including the habitat capacity of the Clearwater Basin, we roughly estimate that the maximum number of spawners which could be supported in the Snake River Basin, is about 200,000 fall chinook. We assumed that 200,000 spawners corresponds to the number of spawners,  $P_m$ , that will result in maximum recruitment,  $R_m$ , for fall chinook. Ricker (1975) shows that

$$\beta = 1/P_m$$

Thus, our estimate of  $\beta$  was  $1/200,000 = 5E-06$ .

## 6.2 ABUNDANCE MEASURES

### 6.2.1 Spawners

The abundance of wild spawners has been indexed by dam counts and redd counts. The nursery area from which these fish originated has changed over time, due to the elimination of access to spawning areas above Brownlee Dam in 1958, Oxbow Dam in 1961, and Hells Canyon Dam in 1967. Mitigation requirements, which are intended to reflect the maximum production capacity of the habitat, were set at 17,800 fall chinook above Oxbow and Brownlee dams, and 6,600 fall chinook between Hells Canyon and Oxbow Dams. Schaller and Cooney (1992) estimate that the remaining habitat, excluding the Clearwater Basin, can support up to 3,800 spawners. Construction of these dams would have caused a decline in the number of wild adults returning through 1970, so only after 1970 can declines in the wild run be assigned entirely to factors downstream of Hells Canyon Dam. Adults that returned to Hells Canyon Dam during 1967-1970 (from juveniles that had reared above the dam) were trapped at the dam and either were released above the dam or were used as broodstock for hatchery purposes, so they would not have caused excessive spawner density below the dam.

Estimates of wild fall chinook passing the uppermost dam in the Snake River are used as an indicator of population trend. Counts began at Ice Harbor Dam in 1962, and the last dam to be completed was Lower Granite in 1975. ODFW (1991) has reported the number of naturally produced chinook escaping above the dams each year (Table 6-2). Counts of naturally produced fish dropped sharply after a high of 19,500 in 1968. The eradication of fall chinook habitat above Hells Canyon Dam in 1967 leaves us with no



spawning areas in the Snake River Basin that can serve as a measure of long term spawner trends in unperturbed habitat, in the way that we used wilderness areas for spring chinook. Therefore, we can only compare the observed rate of decline to that predicted by the model we developed for rebuilding analysis (see Chapter 8). The simulated returns match the observed declining trend in spawners closely.

Table 6-2. Estimates of hatchery and wild fall chinook escapement above Snake River Dams, 1964-1990 (from ODFW 1991).

Year	Ice Harbor	L. Monum.	Little Goose	L. Granite	Lyons Ferry Trap	Hagerman Releases	Lyons Ferry & other hatchery Origin	Hagerman Origin	Percent Hatchery	Estimated Wild Fish
1964	9,100	--	--	--	--	--	--	--	0	9,100
1965	8,200	--	--	--	--	--	--	--	0	8,200
1966	12,800	--	--	--	--	--	--	--	0	12,800
1967	14,000	--	--	--	--	--	--	--	0	14,000
1968	19,500	--	--	--	--	--	--	--	0	19,500
1969	13,600	6,200	--	--	--	--	--	--	0	6,200
1970	9,000	5,300	4,500	--	--	--	--	--	0	4,500
1971	9,300	7,800	4,700	--	--	--	--	--	0	4,700
1972	7,500	4,100	1,800	--	--	--	--	--	0	1,800
1973	6,700	3,800	2,400	--	--	--	--	--	0	2,400
1974	2,400	2,200	900	--	--	--	--	--	0	900
1975	1,900	1,800	900	1,000	--	--	--	--	0	1,000
1976	1,100	1,100	430	470	--	--	--	--	0	470
1977	1,200	870	420	600	--	--	--	--	0	600
1978	1,100	500	490	640	--	139,000	--	--	0	640
1979	1,200	620	550	500	--	365,000	--	--	0	500
1980	1,200	570	500	450	--	181,000	--	--	0	450
1981	770	490	420	340	--	709,000	--	25	0.07	315
1982	1,600	930		720	--	79,000	--	132	0.18	568
1983	1,800	800		540	--	427,000	--	229	0.42	311
1984	1,700	620		640	35	128,000	8	274	0.44	358
1985	2,046	982		691	6	--	1	381	0.55	309
1986	3,104	1,741		784	245	--	59	231	0.37	494
1987	6,788	3,327		951	1,654	--	395	239	0.67	317
1988	3,847	1,647		627	327	--	78	131	0.33	418
1989	4,638	2,018		706	701	--	168	21	0.27	517
1990	3,477			391	586	--	91		0.27	251

-- Project was not operational

In 1990 of the estimate 140 hatchery fish reaching Lower Granite Dam, 49 were removed, transported to Lyons Ferry hatchery for broodstock and thereby prevented from passing upstream of the dam into the natural spawning area. Therefore, 391-49 = 342 adult fall chinook passed Lower Granite, of which 91 were presumed to be hatchery fish, resulting in a total wild run of 342-91 = 251 fish.

### **6.2.1.1. Reconstructed Dam Counts**

Lower Granite Dam adult passage was considered to be wild until 1983. Since that time reconstructed dam counts have been used to estimate the number of wild and natural fish passing Lower Granite Dam (Appendix 3).

The current method of reconstruction is presented below (Personal communication, L. LaVoy, WDF, Wenatchee):

- ▶ The U. S. Army Corps of Engineers operates an adult trap above the viewing window at Lower Granite Dam. A metal tag detector trips the trap so that coded-wire-tagged fish passing the detector are captured. These fish are examined for the hatchery source of the tag; the number of tags are tallied by hatchery source and then totaled over hatchery sources.
- ▶ The total number of tags is then compared to the viewing-window dam count of adipose-fin-clipped passage. The viewing-window count is usually higher. The total coded-wire tag count is divided by the window-count of clipped adipose fins to estimate the proportion of marked fish that were caught.
- ▶ Each hatchery's code total is then divided by the estimated proportion caught to estimate that code's passage number.
- ▶ Each hatchery's code's passage estimate is then divided by the hatchery's marking rate to estimate the hatchery's contribution to the dam count.
- ▶ The estimated contributions from the different hatcheries are then added to obtain the estimated total hatchery contribution.
- ▶ The estimated hatchery's contribution is subtracted from the total dam count to estimate the wild/natural passage.

The method is fairly straight-forward. There are a series of rather standard assumptions associated with mark recovery estimates; i.e.

- ▶ The survival of marked fish is equal to that of unmarked fish within each hatchery brood-year,

- The probabilities of a marked fish being trapped is the same for all codes,
- The rate of tag loss, including the inability to read the tags, is the same for all codes.

Failures of these assumptions would introduce bias into the estimates.

Possible sources of error in trapping are being corrected for by an expansion based on viewing-window count. There are a couple of additional sources of bias. The expansion based on viewing-window counts of adipose-clipped fins would not be large enough if there was any night-shift passage of marked fall chinook. Since fish are being trapped on a twenty-four hour shift, such a bias could impact trend estimates as well as abundance estimates. This and the possible failure of the enumerators to count all marked fish are the only additional biases that we could identify. If these biases exist, then the hatchery component of the run is being under-estimated and the wild component of the run is being over-estimated. Video-taping of fish passing Lower Granite Dam during 1992 demonstrated that 11.7% of adult fall chinook and 9.7% of jack chinook passed the dam during non-counting hours (personal communication, D. Hatch, Columbia River Inter-Tribal Fish Commission, Portland).

If it can be determined that biases do not exist or are small, then the historical record could be used to provide abundance and trend measures. Or if the biases can be measured then the historical record can be adjusted for those biases.

**R E C O M M E N D A T I O N : -**  
adjustments using 24-hour counts from the 1992 field test data for video-imagery conducted by the Columbia River Inter-tribal Fish Commission. This sampling should be continued until the variability in night passage has been well established and an adjustment factor with acceptable confidence limits can be established.

**NOTE:** There is a large discrepancy in the wild passage estimated by WDF (Appendix 3) and those estimated by ODFW (Table 6-1). For example, the wild count estimated in 1990 by WDF was 78 fish, whereas the number estimated by ODFW was 215. The WDF data was used as the data source for NMFS's listing of the fall chinook.

**6.2.1.2** Historic Redd Counts

We could find little in the way of a historic record for consistent redd count surveys over years until 1987. Since then, aerial flights have been conducted on a scheduled basis by the U.S. Fish and Wildlife Service. In Table 6-3 we present a table of redd counts based on two standard aerial surveys with additional enumerations from supplementary aerial survey days in 1990 and 1991 and from observations made from a pontoon and from scuba diving in 1991. Total dam count, total escapement for 1990 and 1991, and reconstructed wild counts are also presented.

Table 6-3. Accumulated redd counts associated with increased sampling effort (Rondorf and Milk, 1992)							
Year	Lower Granite Dam Total Count (escapement to spawning grounds)	Lower Granite Dam Wild Count	2 Aerial Redd counts (a)	(a) + 1 aerial survey (b)	(b) + 6 aerial surveys (c)	(c) + pontoon surveys (d)	(d) + SCUBA surveys
1987	951	253	66				
1988	627	368	57				
1989	706	295	58				
1990	385 (335)	78	29	34			
1991	654 (594)	318	28	32	42	53	62

There is a steady decline in redd count associated with the two-aerial-survey base. This was not true for the total dam count or for the reconstructed wild count. However, a longer time series would be required before a trend could be assessed using redd counts.

**6.2.1.3** Future Redd Counts

For fall chinook, it may not be possible to allocate redd counts to wild/natural spawners based on the proportion of spawned-out female carcasses of wild origin at the spawning site, as we recommended for spring-summer chinook. Accurate spawner assessments will probably not be possible in the main stem of the Snake River where deeper and more turbid water will make it difficult to spot and retrieve spawners. If an accurate spawner assessment is not possible then there are two options:

- allocate the redds according to the proportion of wilds passing Lower Granite Dam and escaping to the spawning ground, or
- intercept all fall chinook at Lower Granite Dam and permit only natural fish to escape in which case all redds would be of natural origin.

The first option has a potential bias if the hatchery and the wild fish are not equally successful in digging redds. There would be no bias under the second option.

### **6.2.2 Juveniles**

Relatively little attention has been given to juvenile Snake River fall chinook. Because of this, limited abundance data are available. In 1991 the Fish Passage Center (FPC, 1992) and The Fish Transportation Oversight Team (FTOT, 1992) began distinguishing between Snake River yearling (spring) chinook and subyearling (fall) chinook migrants collected at transport facilities at Lower Granite and Little Goose Dams. Attempts were made to differentiate yearling and subyearling chinook in the past, but were discontinued after the 1986 season due to unreliable data brought about by releases of hatchery spring chinook (Koski et al, 1987).

Fish passage indices for subyearling fall chinook at Lower Granite Dam totaled 13,900 in 1991 (FPC, 1992) and 5,943 in 1992 (Table S-2). number should be regarded as a tentative estimate). Little Goose Dam juvenile passage was not discussed in the FPC annual report, therefore no passage indices were available. Initial identification of subyearling chinook was based on length and morphological characteristics. Juvenile fall chinook were found to have a smaller eye, wider body, and more silvery appearance than spring chinook smolts. With the passing of the summer months, and as migrants increased in size, FPC personnel relied solely on morphological characteristics. Transportation operations at Lower Granite and Little Goose Dams used the criteria developed by the FPC to distinguish subyearling fall chinook from yearling spring chinook.

Subyearling fall chinook first entered collection facilities at Lower Granite Dam on May 10, 1991. Two noticeable peaks of passage were recorded. The first on June 14, and the second on July 25. Collection facilities at Little Goose Dam first identified subyearling fall chinook entering collection facilities June 14. Passage peaked on July 25. and a relatively high numbers of smolts were collected from late July through early August.

Connor et al.(1992) tracked the migratory behavior of subyearling fall chinook salmon from the free flowing Snake River to Lower Granite Dam using PIT tags in 1992. They found that fall chinook fry emerged later and grew faster than expected, so size criteria had to be redefined. Forty-nine of the PIT tagged fish were recovered at Lower Granite Dam and they were retained for electrophoretic analysis of isozyme types. Forty-six were confirmed fall chinook and three were spring chinook (94%). Thus, the new criteria developed by Connor et al. for distinguishing fall chinook were not without error but were reasonably accurate.

Monitoring done at IDFG traps during the construction of the Hells Canyon complex, specifically the Brownlee dam fish barriers (Bell 1957, 1959, 1960 and 1961 as well as Graban 1964), were reported by Connor, Burge and Miller as revealing a bimodal migration pattern after emergence (Table 6-4). Smolts were recorded as emerging in April and May and emigrating from June through September. These findings agree with the timing recorded by Connor, Burge and Miller (1991). This bimodal migration is also evident in FTOT reports of daily collection at Lower Granite dam and to a lesser degree at Little Goose dam (Table 7 in FTOT 1992).

RECOMMENDATION: Annual indices of juvenile fall chinook abundance leaving the Snake River should be continued. Work should continue to resolve uncertainty in race identification and in trapping efficiency. The relationship between spawner abundance and smolt abundance, as well as factors influencing the relationship, should be established.

Table 6-4. Fork length frequency of juvenile chinook trapped from April 1, 1959 to December 31, 1959 on downstream migrant trapping barges below Brownlee Dam (from Bell 1959).

Length in mm.	Apr.	May	June	July*	Aug.	Sept.	Oct.	Nov.	Dec.
20-24	1								
25-29	3	1							
30-34	15	5							
35-39	24	6							
40-44	2	1							
45-49	4								
50-54	3								
55-59									
60-64		1							
65-69		2	1						
70-74	1	2	2						
75-79		6	1						
80-84		6	2						
85-89		19	6						
90-94	1	8	9						
95-99		4	7						
100-104		1	11						
105-109			17		1				
110-114			37						
115-119		1	91	3					
120-124		1	74	12	1				
125-129	2	1	51	43	29	1			
130-134		1	14	147	133				
135-139			8	246	188	2			
140-144			1	177	142	2			
145-149				67	49	3	3		
150-154			1	5	9	7	6		
155-159				2	1	1	6		
160-164						1	39	2	
165-169						2	76	11	
170-174			1			1	151	17	
175-179							115	31	7
180-184			1				53	28	12
185-189							2	12	5
190-194			1					12	6
195-199							2	5	
200-204					1		1	3	1
205-209								1	

$\bar{x}$  = 41.9    71.1    118.1    138.0    138.1    151.3    172.9    180.9    183.8

n = 56    66    336    702    533    20    454    122    31

\* Approximately 93% of the July catch was made between July 21 and July 31.

### 6.3 MEASURES OF SURVIVAL

#### 6.3.1 **Downstream Passage**

Fisher (1992) used the CRiSP model to simulate survival conditions for outmigrating smolts from the Snake River in 1990 and estimated that the average survival, including transported fish, past all projects was 9.23%, which is half the survival rate that he estimated for spring chinook smolts (18.79%). In order for survival past eight projects to be half of the spring chinook survival (85% per project), we calculated that 78% survival per project (22% loss per project) would produce that result ( $78\%^8 = 13.7\%$  survive vs  $85\%^8 = 27.2\%$  survive). Therefore, we used an average 22% mortality per project (including transported fish) in our model.

Several pieces of evidence support the conclusion that survival of fall chinook is half of that for spring chinook. Collection efficiencies at Lower Granite and Little Goose are about 56% for yearling chinook and 35% for subyearling chinook (Walla Walla District Corps of Engineers 1992). Rieman et al. (1991) estimated that predation by squawfish in John Day Reservoir caused 8%-11% mortality on juvenile salmon and steelhead during April and May, but increased to a high of 61% mortality in August when fall chinook were emigrating. Dawley et al. (1985) report that recovery rates of transported and non-transported juvenile chinook in the Columbia River estuary indicate that the survival rate between the Snake River and the estuary is about half for subyearlings of what it is for yearlings.

**RECOMMENDATIONS: Information on passage survival is critical to understanding of the population dynamics of fall chinook. Procedures should be developed and implemented to estimate mortality of juvenile fall chinook during their outmigration through the Snake and Columbia Rivers.**

#### 6.3.2 **Smolt-to-Adult**

Smolt-to-adult survival rates have not been estimated for wild Snake River fall chinook, so we derived these rates from estimates for fall chinook in the mid-Columbia. The life history and distances traveled by fall chinook in the mid-Columbia are similar to those in the Snake River, and recoveries of CWT's from hatchery fish indicate that ocean



distribution, ocean harvest rates, timing of river entry and river harvest rates are similar between the two groups. Mullan (1990) estimated the smolt-to-adult survival rates were similar between spring and fall chinook in the Wenatchee, and Methow Rivers. He estimated the rate was 2.2% to 8% for fall chinook and 1.6% to 8.1% for spring chinook. Mullan included harvest and dam losses for his estimates of recruits, and his data were all from years after main-stem dams were completed. Thus, Mullan's survival estimates reflect the present-day survival, given seven dams below the Wenatchee River and nine dams below the Methow River. However, Mullan's estimates of smolt abundance for spring chinook (not fall chinook) are based on the assumption that overwinter survival rate from parr to smolt was 50%. Fall chinook do not overwinter in freshwater. When we use our estimate of 30% overwinter survival, then the smolt-to-adult survival for mid-Columbia fall chinook was only 60% (.3/.5) of that for spring chinook.

We needed to estimate smolt-to-adult return rate (SAR) before dams were in place. Since juvenile fall chinook smolts (subyearlings) are smaller than spring chinook smolts, they suffer twice as much downstream mortality as spring chinook smolts (yearlings) when both are passing eight dams (Fisher 1992). Thus, if we were to return to the pre-dam state, the survival improvement for fall chinook would be double that for spring chinook. Therefore,

$$\text{fall chinook SAR} = 2 * 0.6 * \text{SAR for spring chinook}$$

This gives  $120\% \times 10.4\% = 12.6\%$  as the smolt-to-adult survival rate for fall chinook before dams were built. This is the value (12.6%) we used in our population model.

**RECOMMENDATION: Smolt-to-adult survival rate of fall chinook should be estimated directly from estimates of Snake River smolt abundance and adult recruits.**

### **6.3.3 Harvest Rates**

Ocean harvest rates of Lyons Ferry fall chinook have been estimated from CWT recoveries of fish released on-station as subyearlings to average 41% for the 1984 and 1985 broods. These estimates were developed by cohort analysis according to methods described in the Pacific Salmon Commission Technical Committee 1987 Annual Report, Supplement B (PSC 1988). CWT recoveries were corrected for interdam loss and river

harvest (Berkson 1991). Based on the same data, in-river harvest rate averaged 47.6%. These harvest rates are similar to those estimated by the PSC (1990) for mid-Columbia fall chinook from Priest Rapids Hatchery (39% for the 1984 brood and 31% for the 1985 brood). Analysis by PSC (1990) indicates that ocean harvest rates for the 1982-85 broods of upriver bright fall chinook decreased 13% from the average rates during 1976-79, so we assumed it had done the same for Snake River fall chinook. Thus, we used an ocean harvest rate in our simulation modeling of 47% for the 1947-81 broods, and 41% for 1982 and later broods.

We are concerned that estimates of harvest rate based on CWT recoveries of Snake River fall chinook reared at Lyons Ferry Hatchery may be biased by the application of interdam loss rates. Mendel et al. (1992) have demonstrated a high fallback rate of fall chinook at each dam in the Snake River. PSC (1991) uses interdam conversion rates (which includes fallback) to back-calculate the number of fall chinook entering the Columbia River, and this adjustment results in the escapement being multiplied by a factor of about 3.0. This adjustment reduces the calculated harvest rate in the ocean and river. Although the fallback at Snake River dams is a real and substantial phenomenon that must be accounted for, it is likely that chinook homing to the Snake River may fallback at a much lower rate than the overall fallback rate, which is known to include a high proportion of stray fish. Lestelle and Gilbertson (1993) discuss this issue and identify interdam conversion rates as a critical uncertainty in assessing the effects of harvest management options. We strongly agree.

Harvest rates for upriver fall chinook can also be estimated each year from statistics on landings and dam counts (ODFW and WDF 1992). These data confirm that harvest rates in the Columbia River on fall chinook are high (Figure 6-3).

The sharp change in the stock-recruitment relationship for upriver fall chinook following the 1946 brood, as shown by Chapman et al. (1982), indicates that there was probably a sharp change in ocean harvest rate around 1950. It seems likely that ocean harvest rates would have increased after World War II. Chapman et al. (1982) presented data on ocean harvest that indicate about 25% of the adult recruits were caught in the ocean from the 1938-1946 broods, and that this harvest increased to about 55% of adult recruits from the 1947-59 broods.

## Harvest Rates on Upriver Fall Chinook

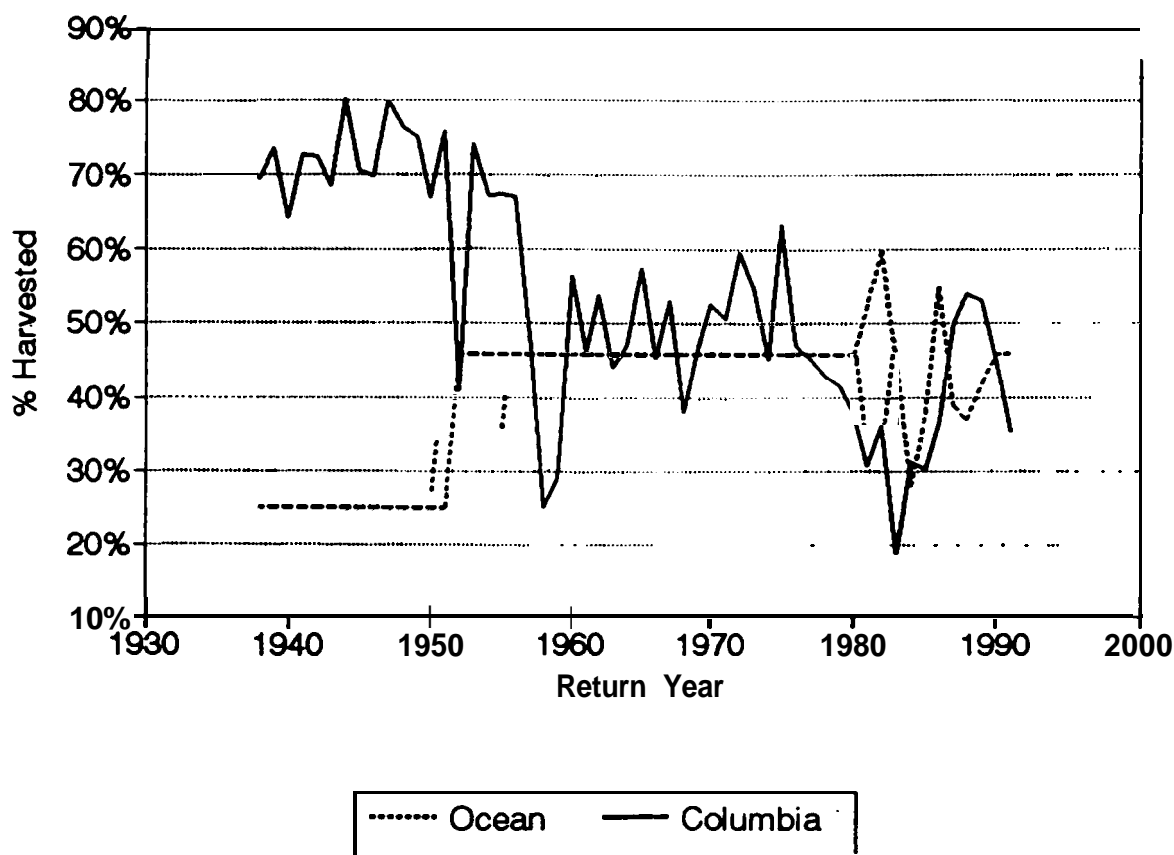
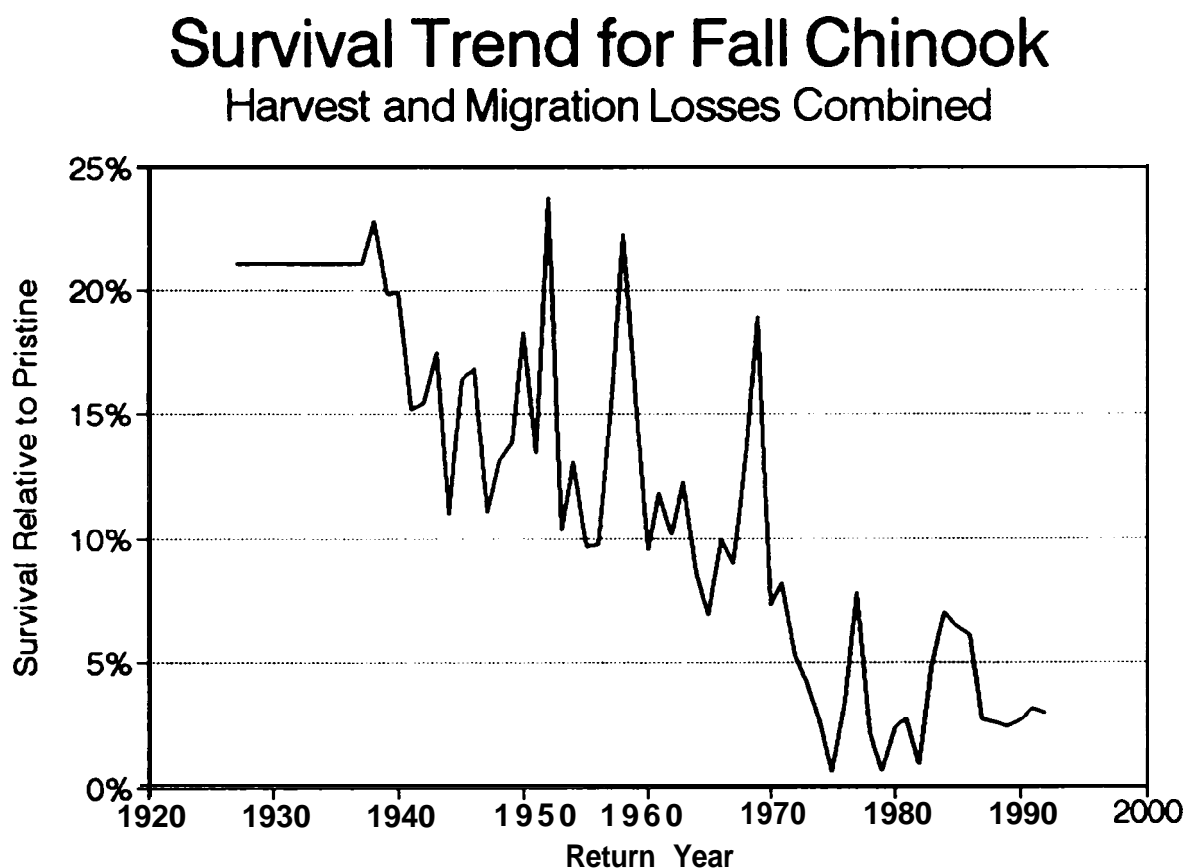


Figure 6-3. Estimated harvest rates in the ocean and Columbia River experienced annually by upriver fall chinook during 1938-1991. See text for data sources.

### 6.3.4 Upstream Passage

We reviewed studies of adult losses between dams in the Columbia River and concluded that the best estimate of adult mortality per dam is 5%. Chapman et al. (1991) estimated an average 4.6% per dam mortality of adult chinook for seven years during 1979-1987 between John Day to Priest Rapids or Ice Harbor. Liscom and Stuehrenberg (1983) radio tracked fall chinook from Bonneville Dam to McNary Dam and concluded that fish count discrepancies were not losses, but resulted from lack of information on tributary escapements, main channel spawners, and harvest. For this reason, we discount estimates of higher loss rates.

The combined effects of high harvest rates, downstream losses and upstream losses increased from the late 1930's to the mid-1970's. The result was a dramatic decline in the survival of fall chinook, particularly when compared to their survival under pristine conditions (no impoundments and no harvest) (Figure 6-4).



**Figure 6-4.** Trend in estimated survival of fall chinook during 1928-1992. Survival is expressed as a percentage of the survival under **pre-dam** conditions with no harvest, and includes the effects of harvest and in-river losses during juvenile and adult migrations.

## 6.4 HERITABLE DEMOGRAPHIC CHARACTERISTICS

### 6.4.1 **Age Composition of Spawners**

Since it will probably not be possible to acquire accurate information from spawning ground surveys for fall chinook, it may be necessary to rely on information from hatchery returns for assessing sex and age distribution. Age three and four fish tend to dominate the age distribution of both yearling and subyearling releases (Schaller and Cooney, 1992).

**RECOMMENDATION: In the future, if Lyons Ferry Hatchery takes natural fish as brood stock, the age and sex distributions should be tallied separately for the natural and hatchery fish. However, the number of natural fish taken may be too small for any precise assessment as to the actual distribution of these demographic characteristics.**

Data are complete for only two broods of CWT fall chinook from Lyons Ferry Hatchery and the age composition of returns from these two broods differed substantially in the proportion of age 2 and 3 spawners. The average age composition of fish spawned at Lyons Ferry Hatchery should not be used for determining age-at-maturity composition for a cohort, because there were many differences between years in the number and age of smolts released from Lyons Ferry, and those differences would have affected the age composition of each run year. While awaiting further age-at-maturity data from CWT groups returning to Lyons Ferry Hatchery, we suggest using the mean age composition of fall chinook returning to Priest Rapids Hatchery for 1982-1986 as a surrogate indicator. The mean age composition for fall chinook at Priest Rapids Hatchery was well within the range of values observed at Lyons Ferry Hatchery.

## 7. DE-LISTING **CRITERIA** FOR SOCKEYE

### 7.1 **STOCK-RECRUITMENT** RELATIONSHIPS AND CARRYING **CAPACITY**

Because the sockeye population is functionally extinct in the wild, it is anticipated there will be at least two generations, or after 1998, before significant numbers of adults will spawn naturally (Bevan et al. 1992). Even if the recovery plan went exceptionally well, it is likely to be 20 years in the future before abundance is sufficiently high that density dependent mechanisms affect the population. Additionally, there is substantial uncertainty regarding the genetic viability of the population, because its origin is uncertain. We believe that speculation as to what the stock-recruitment relationship will be is of little value at this time.

**RECOMMENDATION: A thorough monitoring program for smolt abundance and adult recruits should form the basis for characterizing the stock-recruitment relationship of Redfish Lake sockeye as the stock rebuilds. That relationship should be examined to determine if there is a mortality level beyond which the population declines rapidly, in a similar manner to what has been estimated for spring and fall chinook. If such a mortality level exists, it should be used in some form in the de-listing criteria.**

### 7.2 ABUNDANCE MEASURES

#### 7.2.1 **Spawners**

The abundance of spawners has been indexed periodically by carcass surveys, redd counts, weir counts, and dam counts. Pirtle (1957) reported on the number of observed successful spawners at Redfish lake from 1954-56. Pirtle observed 548 fish in 1954, 1,115 in 1955 and 289 in 1956. Redd counts in Redfish Lake during 1977 through 1989, most of them zero, have been summarized by Hall-Griswold (1990), but should be regarded as unreliable. The redd count records do not indicate lake conditions at the time of the counts, and we found, for example, that the 0 count in 1981 was recorded because heavy wave action on the lake precluded counting (IDFG 1982). Counts of adult sockeye passage at

Redfish Lake weir during 1954-1966 are the only other available indicator of adults reaching the spawning grounds. Counts ranged from a high of 4,365 in 1955 to 11 in 1961, with an average of 657 returns over the 13 year reporting period.

Dam counts of Columbia and Snake River dams may be inaccurate and misleading. For all but five years since 1960 Priest Rapids and Ice Harbor Dam counts added together exceed the total adult sockeye counts reported at McNary Dam (Table 7-1). In fact counts at Priest Rapid dam alone exceed McNary dam counts for all years described by as much as 2.5 times. Several factors have been suggested in explaining these observations. First, adult passage at upper Columbia River dams represent 24 hour counts, where lower Columbia River and Snake River counts have not been conducted at night. In 1992 Bonneville, Ice Harbor and Lower Granite dams have started conducting 24 hour counts (Larry Vasham, FPC personal communication). It is not clear if these attempts have cleared up the irregularity. Fallback may be a second explanation for higher counts upstream of McNary Dam. It could be that Priest Rapids dam experiences high rates of fallback, but it is unclear with present information. Over shooting the spawning ground may or may not contribute to fallback, but again this is not known. A third possible factor might be passage through navigation locks. No information is available to support or refute this theory. Possible influence from kokanee/sockeye that may not of migrated to sea, and possibly matured in-river have not been explored.

RECOMMENDATION: The Recovery Plan should include development of a method of distinguishing ocean adult returns from those that mature in reservoirs downstream of Redfish Lake. It may be necessary to sample scales from all returning adults and subject the scales to strontium analysis to distinguish ocean migrants. (personal communication, C. Knudsen, WDF, Olympia).

More sockeye have been counted passing Lower Granite Dam in some years than were counted passing Ice Harbor in the same year. Causes would be similar to those discussed above. Some Redfish Lake weir counts have also exceeded those reported at Lower Granite Dam.

Table 7-1. Counts of sockeye salmon passing selected dams in the Columbia and Snake River drainages. Counts taken from Broderic (1990) and Mullan (1986).

	BONNEVILLE	M McNARY	PRIEST RAPIDS	ICE HARBOR	ROCK ISLAND	REDFISH WEIR
YEAR						
1938	75,000				17,000	
1939	73,400				19,600	
1940	148,800				26,900	
1941	65,700				900	
1942	55,500				16,300	
1943	39,800				17,700	
1944	15,100				4,900	
1945	9,500				7,100	
1946	74,400				46,600	
1947	171,100				79,800	
1948	131,500				84,600	
1949	51,400				18,700	
1950	77,993				50,100	
1951	169,428				102,700	
1952	184,645				113,700	
1953	235,215				156,000	
1954	130,107	108,181			91,200	598
1955	237,748	173,758			155,800	4,365
1956	156,418	102,145			92,200	1,381
1957	82,915	85,460			71,300	523
1958	122,389	102,397			97,900	55
1959	86,560	83,977			72,300	290
1960	59,713	55,372	58,210		60,300	75
1961	17,111	16,388	19,793		19,200	11
1962	28,179	29,372	28,575	38	29,300	39
1963	60,319	59,744	64,833	1,118	64,700	395
1964	99,856	83,931	79,072	1,276	69,400	335
1965	55,125	42,052	48,340	317	42,400	17
1966	156,661	173,028	170,071	278	164,600	61
1967	144,158	105,635	123,786	717	119,800	1,000*
1968	108,207	101,007	108,308	1,165	104,800	1,500*
1969	59,636	29,787	39,240	745	38,000	1,400*
1970	70,762	59,636	77,419	797	74,900	1,000*
1971	87,477	52,867	73,837	532	71,400	1,000*
1972	56,323	26,422	44,927	363	43,500	1,100*
1973	56,979	42,731	54,480	233	68,700	100*
1974	43,837	26,505	35,434	204	33,900	300*
1975	58,212	43,143	55,210	243	54,400	200*
1976	43,611	24,632	32,810	771	35,400	700*
1977	99,829	80,781	95,413	582	90,300	600*
1978	18,436	18,511	17,529	86	14,700	200*
1979	52,628	37,792	45,662	30	50,500	100*
1980	58,882	44,301	52,039	36	52,658	100*
1981	56,037	26,644	51,456	142	47,139	26*
1982	50,219	15,077	40,461	174	41,111	50*
1983	100,545	40,903	90,008	216	86,424	0*
1984	152,541	56,905	114,761	105	109,074	22*
1985	166,340	98,457	118,542	24	103,230	
1986	58,123	46,443	43,084	20	49,789	
1987	116,993	72,194	76,578	13	70,673	
1988	79,914	50,080	51,135	22	49,159	29
1989	41,879	41,318	45,301	2	37,360	



Chapman et al. (1990) attempted to reconstruct escapement to Redfish Lake by subtracting dam counts at Rock Island Dam from counts at McNary Dam (including estimates of 2-3% per dam for inter dam loss). These attempts revealed inconsistencies, and were abandoned. Chapman et al. (1990) also made attempts to estimate Snake River escapement from Rock Island and Bonneville dam counts, adjusting for in-river harvest and inter-dam loss, but this approach was similarly unsuccessful.

Estimates for adults entering Redfish Lake were also attempted by Mullan (1986) and Broderic (1990), but appear to be higher than dam counts at Ice Harbor Dam (Table 7-2). Chapman was able to compare Ice Harbor Dam counts and Redfish weir counts from 1962-1966. In the first year of Ice Harbor Dam counts one less fish was recorded than was observed at Redfish Lake weir (38 at IHD and 39 at Redfish Lake). The following four years counts at Redfish Lake were between 35-50% of the total adult counts at Ice Harbor Dam, which may suggest high mortality between the two sites. Other factors, such as fallback over Ice Harbor Dam, may contribute to these low percentages of survival.

### **7.2.2 Juveniles**

The potential for bias in past measures of juvenile sockeye abundance is substantial and the problems associated with this potential far override any concerns about precision of the measures. The major issue, "How do you distinguish a kokanee smolt from a sockeye smolt?"

Sockeye at Redfish Lake Creek weir were studied from 1955-1964 by Bjornn, Craddock and Corley (1968). Estimates of emigrating smolts ranged from 2,133 (1960) to 65,000 (1957). Chapman et al. (1990) took these estimates and arrived at an average number of 28,757 smolts originating from above the Redfish Lake Creek Weir.

Chapman et al. (1990) estimated smolt numbers passing the upper most dam on the Snake River in the 1980's by adjusting sockeye smolts collected at Little Goose and Lower Granite dams (Table 7-2). Mean smolt yield from Redfish Lake for 1981-1989 as estimated by Chapman et al. (1990) did not differ significantly from estimates in 1955-66.

Table 7-2 Estimated smolt abundance and survival to adult return to the mouth of the Columbia River of Redfish Lake sockeye (from Chapman et al. (1990), as adapted from Bjornn et al. (1968).

Smolt year	Smolt <sup>a</sup> number	Adults <sup>b</sup> to weir or dam	Survivals				
			Weir:weir <sup>c</sup>	Interdam <sup>d</sup>	Fishery <sup>e</sup>	Weir: Col.R. <sup>f</sup>	L.Gran: Col.R. <sup>g</sup>
1955	54,000	523	0.97%	94%	56.0%	1.84%	2.10%
1956	38,029	55	0.14	94	37.1	0.40	0.47
1957	65,000	290	0.45	91	31.7	1.56	1.81
1958	41,000	75	0.18	91	33.6	0.59	0.69
1959	13,000	11	0.08	91	30.1	0.29	0.34
1960	2,133	39	1.83	91	63.0	3.19	3.71
1961	21,600	395	1.83	88	78.6	2.65	3.08
1962	23,000	335	1.46	88	80.2	2.07	2.41
1963	23,320	17	0.07	86	89.4	0.09	0.10
1964	6,492	61	0.94	86	97.5	1.12	1.30
mean							1.60
45% FGE:							
1981	14,509	122	0.83	78	96.7		1.11
1982	26,651	49	0.18	78	79.1		0.30
1983	11,898	35	0.29	78	59.1		0.64
1984	41,680	15	0.05	78	89.8		0.05
1985	14,371	29	0.20	78	53.3		0.49
1986	16,467	23	0.14	78	51.3		0.35
1987	18,137	2	0.01	78			0.02
1988	8,293	---					
1989	31,211	---					
1990	32,693 <sup>h</sup>						
mean							0.42
33% FGE:							
1981	19,785	122	0.62	78	96.7		0.82
1982	36,342	49	0.13	78	79.1		0.22
1983	16,222	35	0.22	78	59.1		0.47
1984	52,781	15	0.03	78	89.8		0.04
1985	19,597	29	0.15	78	53.3		0.36
1986	22,455	23	0.10	78	51.3		0.26
1987	24,445	2	0.008	78			0.01
1988	10,616	---					
1989	39,882	---					
1990	44,582 <sup>h</sup>						
mean							0.31

a - Smolts at weir 1955-64, smolts at Lower Granite 1981-90. Latter calculated as footnote C, Table 3.

b - Adults at weir for smolt years 1955-64, and at Lower Granite for 1981-88.

c - Redfish weir to Redfish weir 1955-64, Lower Granite Dam to Lower Granite Dam 1981-87.

d - Adult inter-dam survival, with 3% inter-dam loss per project.

e - Zones 1-6.

f - Redfish weir to return to mouth of Columbia River.

g - Smolt numbers at Redfish weir, adjusted by 86% survival to Lower Granite damsite. 1955-64. For all years survival is from Lower Granite damsite to adult return to Columbia River.

h - To June 21, 1990. Probably represents close to 95-100% of eventual total.

Chapman divided smolt collections at Lower Granite dam by fish guidance efficiencies (FGE) of 45% and 33%. FGE of 33% was derived from looking at the relationship of FGE at Priest Rapids Dam between chinook and sockeye migrants (58% greater efficiency for chinook passage than sockeye), and assuming that FGE at Lower Granite dam for sockeye was also 58% of the 57% FGE estimated there for chinook. Chapman et al. (1990) assumed an average project loss of 15% (based on chinook yearling data from FPC 1987, McKenzie et al. 1983 and 1984). This may be biased since sockeye have been shown to pass through intakes deeper in the water column than other anadromous smolts (Olson 1984, Brege et al. 1988). Also, FGE at Lower Granite tends to be lower than desired for chinook (Swan et al. 1990), therefore sockeye FGE may be similarly effected.

The Fish Passage Center has recorded juvenile sockeye passage since 1984. Between the years of 1984 and 1991 sockeye smolt passage indices for Lower Granite Dam have ranged from a low of 700 sockeye smolts in 1987 to a high of 25,900 in 1991. We find that these indices are unreliable for Redfish Lake sockeye, because of uncertainty regarding their origin and imprecision in the sampling methods. It is not known how many of these smolts are anadromous sockeye destined for the Pacific Ocean. Through studies conducted by IDFG, it has been determined that populations of Dworshak Reservoir kokanee have been passing through the turbines and over spillways at Dworshak Dam. The degree to which Dworshak reservoir kokanee have influenced Snake River sockeye passage indices is unknown.

Fish Passage Center and FTOT data are an index of guided or collected fish only and are not adjusted for factors other than flow. The index is divided by the proportion of flow through the powerhouse.

The fact that collections of sockeye smolts at Little Goose Dam have peaked before they did at Lower Granite raises some serious questions about the efficiency of detecting sockeye migrants at collection facilities. It has been reported (Chapman et al. 1990, Olson 1984, Brege et al. 1988) that sockeye travel deeper in the water column and are more likely to pass through the turbines than are chinook.

Percent of age class data of emigrating sockeye (Bjornn et al. 1990) was given showing age 2 sockeye predominating in 1956, and from 1958 to 1960 (between 63.8 and 97.9% of the migrants) changing to age 1 sockeye predominating from 1961-1964 and 1966 (between 66.5 and 98% of the migrants). In 1957 and 1965, age class distribution was

relatively equal (1957; 59.9% I to 40.1% II, and in 1965; 46.2% I to 53.8% II).

### 7.3 MEASURES OF SURVIVAL

#### 7.3.1 ~~Egg-to-Smolt~~

Chapman et al. (1990) estimated egg-to-smolt survival of Redfish Lake sockeye for the years of 1979-1988 (Table 7-3). These estimates are the best available, but should be considered as rough guesses, because of the many assumptions required to derive the estimates. Lower Granite Dam adult escapement was used to estimate the number of eggs deposited where a 1:1 sex ratio (Bjornn et al. 1968) was assumed. The number of females was reduced by 25% to allow for pre-spawning mortality (from Warren 1988). The fecundity of each spawning female was assumed to be 2,430 eggs (Warren 1988). Smolt abundance at Redfish Lake was backcalculated from FPC smolt collections at Lower Granite Dam, where collections were divided by an FGE of 0.45 to arrive at an estimated number of smolts reaching Lower Granite Dam (FPC collections at Little Goose Dam were used in 1984, 1987 and 1988, because collections exceeded those experienced at Lower Granite. Smolts were divided by FGE of 0.45 and a per project mortality of 0.85. The number of smolts collected at Lower Granite Dam were then added to estimate the total number arriving at Lower Granite Dam). Number of smolts at Redfish Lake were estimated by adjusting for a 5% reservoir mortality rate and 140% mortality between Redfish Lake and Lewiston (based on Bell et al. 1976: 0.7% loss per day with a migration rate of 22 miles a day). All migrants were assumed to be age 1.

Chapman et al. (1990) concluded that their smolt:egg survival was much too high to attribute the bulk of smolt emigrants from Redfish Lake to egg deposition by adult sockeye alone. Bjornn et al. (1968) estimated a mean egg-to-smolt survival of only 20% (1955-68) as opposed to Chapman's mean of 70%. Based on this difference, Chapman et al. (1990) suggests that most of the smolt yield from Redfish Lake are contributed by kokanee stocks. We conclude that these estimates of survival are not a sound basis for comparing past to future population performance. Thus, such estimates should not be included as de-listing criteria.

Table 7-3. Estimated egg:smolt survival of sockeye salmon from Redfish Lake (from Chapman et al. (1990)). See text for estimation methods.

1	2	3	4	5	6	7
<u>Adult</u> <u>year</u>	<u>L.Gr.</u> <u>count</u>	<u>Female</u>	<u>Egg</u> <u>depos.</u> <sup>b</sup>	<u>L.Gr.</u> <u>smolts</u> <sup>c</sup>	<u>Redfish</u> <u>smolts</u> <sup>d</sup>	<u>Smolt:</u> <u>egg surv.</u> <sup>e</sup>
1979	25	13	23,693	14,509	17,759	0.75
1980	96	48	87,480	26,651	32,304	0.37
1981	218	109	198,652	11,898	14,422	0.07
1982	211	105	191,363	41,680	50,521	0.26
1983	122	61	111,173	14,371	17,419	0.16
1984	49	24	43,740	16,467	19,960	0.46
1985	35	17	30,983	18,137	21,984	0.71
1986	15	7	12,758	8,833	10,707	0.84
1987	29	15	27,338	31,211	37,832	1.38
1988	23	11	20,048	32,693	40,016	2.00
			Means =		26,292	0.70

- A - Assumed 50% females (Bjornn et al. 1968).
- B - Females adjusted for 25% pre-spawn mortality, and multiplied by fecundity of 2,340 (Warren 1988).
- C - Smolt collection at Lower Granite Dam, divided by 0.45 for assumed fish guidance efficiency. Where Little Goose collection exceeded Lower Granite collection (1984, 1988), we used the Little Goose number, divided by assumed fish guidance efficiency of 0.45 and project mortality of 0.85, then added smolts collected at Lower Granite Dam to estimate numbers of smolts that arrived at Lower Granite Dam.
- D - Lower Granite smolts adjusted for 5% reservoir mortality in Lower Granite pool and 14% mortality between Redfish Lake and Lewiston (assumed 0.7% loss rate per day (Bell et al. 1976) and a migration rate of 22 miles per day in the free-flowing river segments). Resulting divisor is 0.825.
- E - Assumes all smolts leave Redfish Lake at age I. Absence of length frequencies mandates this assumption. Presence of substantial numbers of age II smolts would not change conclusions drawn from this table.

### 7.3.2 Downstream Passage

Downstream passage loss of sockeye smolts has not been directly measured, but the loss has been assumed to similar to the 15% level estimated for yearling chinook smolts (McKenzie et al. 1983 and 1984). This may be biased since sockeye have been shown to pass through intakes deeper in the water column than other anadromous smolts (Olson

1984, Brege et al. 1988). Survival estimates should be completed for mid Columbia sockeye stocks and used as a surrogate for Snake River sockeye until data can be obtained directly from Redfish Lake stock.

### **7.3.3 Smolt-to-Adult**

Chapman et al. (1990) estimated smolt to returning adult survival for sockeye at lower Granite Dam to the Columbia River (1981-87). These estimates ranged from 0.02% to 1.11% (mean 0.42%) if FGE was assumed to be 45%, and 0.01% to 0.82% (mean 0.31%) if FGE was assumed to be 33% (see Table 7-3). Chapman then took data on smolt abundance collected by Bjornn (1955-64) and adjusted the smolt estimates upward to account for 86% survival at Lower Granite Dam. From these calculations, estimated smolt to returning adult survival ranged from 0.10% to 3.71% (mean 1.60%). These estimates assumed a typical migrant spent two years in the ocean. The estimates represented survival to the mouth of the Columbia River because they were adjusted for interdam loss and gill net fisheries. Missing data from the late 1960's and 1970's does not allow for the development of a continuous trend, but it is clear that survival rates were lower in the 1980's than they were in the mid 1950's and early 1960's.

Bjornn et al. (1968) estimated that smolt-to-adult survival for returns to Redfish Lake weir ranged from 0.07% to 1.83% with a ten year mean of 0.79%. Chapman et al. (1990) used Bjornn et al.'s estimates to calculate survival to the mouth of the Columbia River. Chapman et al. adjusted Bjornn's estimates for in-river harvests (mean harvest rate of 40.3%) and interdam loss of adults (3% correction per dam). The resultant survival rates to the mouth of the Columbia River ranged from 0.09% to 3.19% (mean 1.38%).

### **7.3.4 Harvest Rates**

Essentially all harvest of Snake River sockeye occurs in the Columbia River. Pratt and Chapman (1989) reviewed CWT mark data on mid-Columbia sockeye and found no ocean recoveries from 24 tag groups from the 1976-78 smolt years, and 72 tag groups during the 1982-85 smolt years. High seas tagging of sockeye done by Fredin et al. (1977) was reported by French et al. (1976) to produce only one recovery (in 1962) in the Columbia River (from Pratt and Chapman 1989).

Exploitation rates of sockeye in the Columbia River have varied from 0 to 86% between 1938-1990 (ODFW and WDF 1991). We assume that harvest rates are similar for mid-Columbia and Snake River stocks. No Fisheries targeting sockeye in zones 1-5 took place from 1977-1983, but directed sockeye fisheries (including Zone 6) were conducted from 1983-88. Table 7-2 shows that these harvests took up to 49% of the sockeye entering the river as recently as 1986. Landings by ceremonial and subsistence fisheries during 1977-1991 ranged from less than 100 in 1986-88 to 3,300 in 1991 (ODFW and WDF 1991), which represents 0 to 4.3% of the run.

### **7.3.5 Upstream Passage**

Chapman et al. (1990) suggested an inter dam loss of 2-3% per project, which would equate to 78% survival past all dams under present conditions.

## 8. **SIMULATION** OF STOCK REBUILDING

Our objective in simulating stock rebuilding was to evaluate the influence of several population parameters on population growth and to determine the overall improvements in survival rates (regardless of conservation actions used to achieve them) that would be required to meet the specific de-listing criteria within specified time periods of 5 years, 10 years, and 50 years. In order to conduct this evaluation, we developed a simplified life-cycle model that includes the stock-recruitment functions and survival rates described in this report. Several population models have been developed and are in use by resource agencies in the Columbia Basin, but we found them far more complicated than needed to answer the simple question we wanted to answer. In the process of developing our own streamlined model, it became obvious that the accuracy of information on survival rates and stock-recruitment relationships was insufficient to accurately predict the number of years required for population recovery. Therefore, we used the simulations to identify ranges of recovery times that might be expected to achieve the recommended de-listing criteria. A description of the models we used for each population follows.

### 8.1 SPRING/SUMMER CHINOOK

#### 8.1.1 **Model Description**

The model begins with the number of adults that survive to spawn (Figure 8-1). The number of smolts that these spawners produce in the next generation is predicted according to a Ricker stock-recruitment function. These smolts are then assigned a survival rate to the time that they become adults in the ocean. The survival rate is intended to reflect survival before main-stem dams were built. We use 10.4% as the pre-dam value for smolt-to-adult survival as described earlier in this report. In our annual accounting of fish production and harvest for hindcasting simulations, we use Raymond's (1988) empirical estimate of smolt-to-adult survival for each year and we scale his estimates to the appropriate interdam loss rates of fish migrating upstream and downstream. During the years before 1964, we use the 1964-68 average, and in the years after 1984, we use the 1981-1984 average. Once smolts are converted to adult recruits, they are divided into groups that matured after either 2 yr in the ocean or 3 yr in the ocean. Harvest in the ocean is assumed to be negligible. A portion of the maturing fish entering the Columbia River



each year are harvested in the river. A portion of the adults that are not harvested are lost at each dam. Fish that survive the main-stem harvest and upstream passage are then subject to harvest in the subbasin and prespawning mortality before they spawn. At spawning, the life cycle is complete and the next generation begins.

The parameter values we used in this model are as follows:

Parr Carrying Capacity	25.3 million parr
Parr-to-smolt Survival (overwinter)	30%
Ricker $\alpha$	10
Ricker $\beta$ (spawner to smolt)	$4.7 \times 10^{-6}$
Smolt-to-Adult survival rate (pre-dam)	10.4%
Smolt mortality per dam	15%
Proportion of adults 2-salt	30%
Proportion of adults 3-salt	70%
River Harvest Rate	Annual values
Adult mortality per dam	5%
Prespawning Mortality Rate	20%

### 8.1.2 Simulation Results

#### 8.1.2.1 Hindcasting

Initially, we simulated the population during 1962-1992 to evaluate how closely the simulated spawner abundances matched the observed abundance. The model was not intended to predict abundance, but rather to reflect trends in abundance, so that rebuilding rates could be estimated. We applied harvest rates each year equal to those reported by WDF and ODFW (1992) for upriver spring chinook (Table 8-1). Simulated spawner abundance, when the population was treated as one unit, started with a much higher escapement and dropped more precipitously than the observed counts at the uppermost Snake River dam (Figure 8-2). This result is somewhat of a paradox, because the high initial population indicates the recruitment function is too productive, but the exaggerated decline in the simulated population during the 1970's indicates that either the Ricker  $\alpha$  is too small or the assumed mortality rates per dam are too high. Both of these deductions may be true. **Additionally**, we believe the lack of accounting for differences in productivity between population subunits is contributing to the mismatch of observed and predicted numbers of spawners.

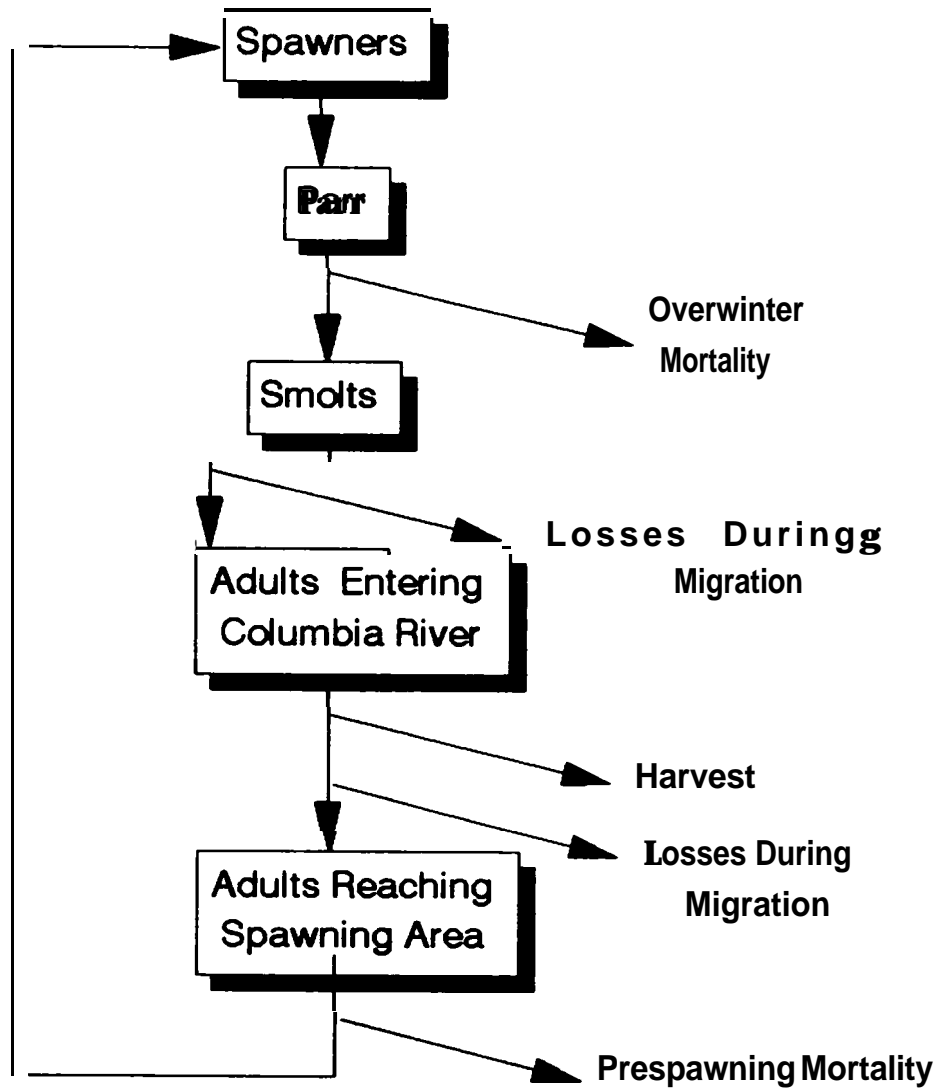


Figure 8-1. Flow diagram of the population model used to evaluate rebuilding schedules for spring/summer chinook.

Table 8-1. Simulated spawner abundance and the annual harvest and survival rates used in the simulation model for spring chinook.

Brood Year	Spawners	Smolts	Return Year	In-river Harvest Survival	Dam Survival Factor	Adults Above Dams					Sum of Subunits
						Population Pooled	Excellent Habitat	Good Habitat	Fair Habitat	Poor Habitat	
1930	235,000	7,550,861	1934	39.4%	100%	200,000	15,000	100,000	130,000	15,000	260,000
1931	235,000	7,550,861	1935	39.4%	100%	200,000	15,000	100,000	130,000	15,000	260,000
1932	235,000	7,550,861	1936	39.4%	100%	200,000	15,000	100,000	130,000	15,000	260,000
1933	235,000	7,550,861	1937	39.4%	100%	200,000	15,000	100,000	130,000	15,000	260,000
1934	235,266	7,549,950	1938	39.4%	95%	200,000	23,354	117,591	87,011	7,428	235,385
1935	235,258	7,549,979	1939	39.7%	95%	200,000	22,596	117,444	85,214	6,931	232,185
1936	236,892	7,544,181	1940	58.1%	81%	200,000	21,085	117,927	81,417	5,745	226,174
1937	331,105	6,768,239	1941	43.1%	81%	200,000	30,096	164,857	112,765	7,579	315,298
1938	211,822	7,590,505	1942	31.7%	81%	200,000	19,014	105,491	75,544	4,840	204,890
1939	149,568	7,184,093	1943	42.6%	81%	200,000	12,850	74,488	57,041	3,697	148,076
1940	213,624	7,590,436	1944	33.6%	81%	200,000	20,915	106,383	67,778	3,807	198,883
1941	164,778	7,367,951	1945	39.9%	81%	200,000	16,467	82,059	49,765	2,614	150,804
1942	201,700	7,580,410	1946	47.9%	81%	200,000	19,220	100,444	59,984	2,908	182,567
1943	239,159	7,535,553	1947	64.2%	81%	200,000	23,806	119,099	65,896	2,872	211,673
1944	326,403	6,821,406	1948	14.3%	81%	200,000	30,441	162,545	96,585	4,081	293,652
1945	70,436	4,909,708	1949	29.6%	81%	200,000	6,020	35,076	23,337	991	65,424
1946	124,317	6,724,677	1950	37.1%	81%	200,000	10,833	61,908	44,142	1,946	118,829
1947	135,921	6,961,639	1951	39.3%	81%	200,000	14,895	67,687	33,547	1,135	117,265
1948	170,630	7,422,377	1952	33.0%	81%	200,000	18,208	84,972	43,612	1,492	148,288
1949	149,452	7,182,447	1953	61.6%	77%	200,000	15,683	74,425	33,760	941	124,810
1950	288,968	7,202,435	1954	52.4%	77%	200,000	29,046	143,903	67,966	1,865	242,780
1951	240,579	7,529,811	1955	20.2%	65%	200,000	23,252	119,806	59,184	1,459	203,701
1952	85,475	5,550,940	1956	24.1%	65%	200,000	6,966	42,566	27,063	719	77,234
1953	86,060	5,573,541	1957	53.4%	62%	200,000	8,037	42,857	23,792	550	75,237
1954	153,068	7,232,109	1958	36.1%	62%	200,000	16,766	76,226	34,257	686	127,935
1955	113,021	6,447,436	1959	44.0%	53%	200,000	12,486	56,283	23,631	412	82,812
1956	152,085	7,218,975	1960	51.6%	53%	200,000	16,088	75,737	31,013	496	123,334
1957	143,442	7,091,374	1961	60.1%	53%	200,000	15,910	71,433	26,455	355	114,152
1958	179,675	7,490,123	1962	43.7%	50%	200,000	18,712	89,476	34,406	428	143,622
1959	131,276	6,872,334	1963	45.0%	50%	200,000	13,521	65,374	24,452	260	103,608
1960	136,834	6,978,352	1964	51.2%	43%	200,000	13,762	68,142	26,834	268	109,008
1961	130,377	6,854,198	1965	40.1%	43%	200,000	14,095	64,926	22,784	189	101,994

Table 8-1. Continued

Brood Year	Spawners	Smolts	Return Year	In-river Harvest Survival	Dam survival Factor	Adults Above Dams					Sum of Subunits
						Population Pooled	Excellent Habitat	Good Habitat	Fair Habitat	Poor Habitat	
1962	90.381	<b>5,735,573</b>	1966	66.0%	34%	<b>163,210</b>	<b>9,730</b>	45.009	15.960	<b>122</b>	<b>70,820</b>
1963	130.568	<b>6,858,077</b>	1967	52.5%	36%	<b>24,917</b>	<b>14,580</b>	65.021	20.686	<b>131</b>	<b>100,419</b>
1964	<b>75,934</b>	<b>5,157,759</b>	1968	63.6%	38%	<b>28,853</b>	<b>8,850</b>	37.814	11,070	<b>60</b>	<b>157,794</b>
1965	103.082	6.162.036	1969	65.6%	59%	<b>152,009</b>	11.804	<b>51,334</b>	14.512	<b>68</b>	<b>77,718</b>
1966	121.679	6.664.197	1970	59.4%	26%	<b>124,335</b>	15.125	60.595	14.424	<b>53</b>	<b>90,187</b>
1967	139.644	7.028.116	1971	68.6%	35%	<b>102,381</b>	16.385	89.541	16.891	<b>57</b>	<b>100,874</b>
<b>1968</b>	114.041	6.474.469	1972	52.6%	26%	<b>101,309</b>	12.569	<b>56,791</b>	13.298	<b>35</b>	<b>82,086</b>
1969	94.647	5.886.916	1973	48.9%	19%	<b>102,329</b>	10.224	47.133	11,323	<b>27</b>	<b>66,708</b>
1970	55.863	4.170.325	1974	64.0%	10%	<b>105,374</b>	<b>6,436</b>	27,819	5.976	<b>11</b>	<b>10,242</b>
1971	<b>44,507</b>	<b>3,504,966</b>	1975	99.8%	3%	<b>11,322</b>	5.417	22.184	4.434	<b>8</b>	<b>12,023</b>
1972	25.296	<b>2,180,744</b>	1976	99.5%	<b>12%</b>	<b>11,222</b>	3.481	12.598	1,995	<b>3</b>	<b>18,077</b>
1973	13.035	<b>1,190,382</b>	<b>1977</b>	70.5%	28%	<b>12,270</b>	2.080	6.491	<b>803</b>	<b>1</b>	<b>9,375</b>
1974	17.016	1.525.131	1978	94.2%	8%	<b>12,325</b>	3.134	8.474	814	<b>1</b>	<b>12,422</b>
1975	21.543	<b>1,890,123</b>	1979	95.9%	2%	<b>15,372</b>	<b>4,508</b>	10.728	819	<b>1</b>	<b>18,056</b>
1976	7.670	716.312	1980	97.0%	8%	<b>15,329</b>	1,752	3.819	231	<b>0</b>	<b>5,902</b>
1977	3,839	366.139	1981	93.8%	9%	<b>15,329</b>	<b>996</b>	1.912	88	<b>0</b>	<b>2,985</b>
1976	3,838	366.001	1982	<b>91.7%</b>	4%	<b>15,329</b>	<b>1,172</b>	1,911	68	<b>0</b>	<b>3,152</b>
1979	2.147	206.388	1983	93.8%	12%	<b>15,329</b>	761	<b>1,069</b>	29	<b>0</b>	<b>1,859</b>
1980	1,339	129.161	1984	92.0%	16%	<b>15,329</b>	<b>580</b>	<b>667</b>	<b>13</b>	<b>0</b>	<b>1,260</b>
1981	1.748	168.318	<b>1985</b>	95.0%	16%	<b>15,329</b>	<b>884</b>	<b>870</b>	13	<b>0</b>	<b>1,787</b>
1962	1,761	169.581	1986	<b>92.0%</b>	23%	<b>15,329</b>	<b>1,090</b>	<b>877</b>	10	<b>0</b>	<b>1,976</b>
1983	2.379	228.388	1987	92.2%	18%	<b>15,329</b>	1,726	1.185	10	<b>0</b>	<b>2,921</b>
1904	3,070	293.785	1986	86.7%	18%	<b>15,329</b>	2.591	<b>1,529</b>	9	<b>0</b>	<b>1,129</b>
1985	3,233	309.143	1989	<b>90.7%</b>	18%	<b>15,329</b>	<b>3,066</b>	1.610	7	<b>0</b>	<b>4,683</b>
1966	4.071	<b>387,779</b>	1990	91.3%	18%	<b>15,329</b>	4.219	2.027	<b>7</b>	<b>0</b>	<b>6,253</b>
1987	4.571	434.390	1991	91.3%	18%	<b>15,329</b>	7.611	3.431	9	<b>0</b>	<b>11,051</b>

## Snake River Returns of Spring Chinook

Simulated vs. Observed Wild/Natural

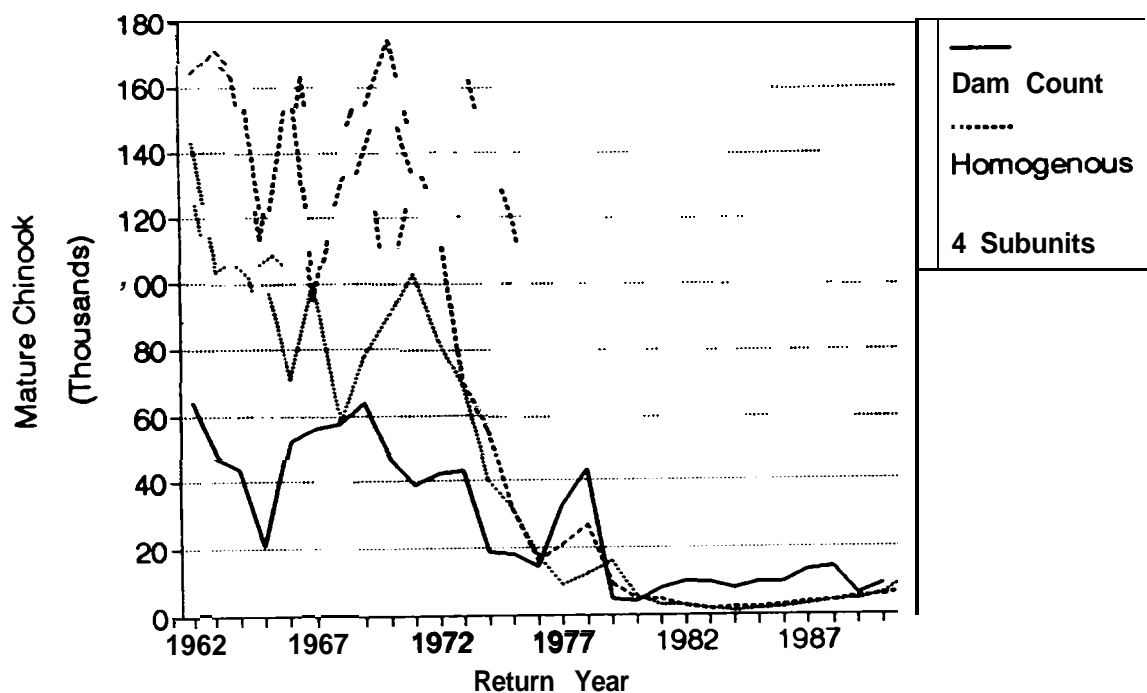


Figure 8-2. Simulated abundance of spring chinook spawners in the Snake River Basin, 1962-1992, compared to counts of wild/natural spring and summer chinook at the uppermost Snake River Dam (indicated as Ice Harbor). Estimates of wild/natural escapement from Petrosky (1991).

We repeated the simulations with the four population subunits corresponding to habitat quality, as described in Section 5.1.6 (Table 8-1). These simulations provided a better fit to the observed trends, but still overreacted to the mortality rates that we assumed were imposed by dams (Figure 8-2). It appears probable from the comparison of predicted to observed trends that a greater proportion of the true population than we assumed may be from the most productive (greater  $\alpha$  values) subunits, and that the overall capacity of the subunits is less than we assumed. These differences highlight the uncertainty that exists regarding the productivity and the carrying capacity of these stocks. However, the trends in our simulations are close enough to the observed trends that the simulation can be instructive for assessing rebuilding rates.

**RECOMMENDATION: Given the uncertainty in the stock-recruitment functions, it is essential that the Recovery Plan emphasize a monitoring system that will enable feedback and correction of the stock-recruitment relationships as rebuilding progresses. This will require 1) estimates of successful spawner abundance, 2) estimates of smolt survival past all eight dams, 3) estimates of harvest, and 4) estimates of adult losses at dams.**

#### 8.1.2.2 Forecasting

We used the four subunit model to simulate rebuilding rates and evaluate the effects of recommended de-listing criteria. We began by simulating the growth of the entire population (summation of the four subunits) over the next 50 years, for four different levels of assumed mortalities: (1) baseline conditions, using 8.7% in-river harvest rate, (2) a 20% improvement in survival (3) a 40% improvement in survival, and (4) a 75% improvement in survival. These levels of improvement are expressed as a percentage of the existing survival, rather than as an absolute number of percentage points. We chose to use the former because management actions will influence survival in a proportionate manner. Baseline survival rate was estimated to be 1.7% from homestream smolt to an ocean adult (10.4% pristine smolt-to-adult survival x 18% passage survival x 91.3% harvest survival). Harvest and survival rates were held constant over the period of the simulation.

These simulations indicated that even under baseline conditions, the population would increase slightly, and that each increment of improved survival would allow a larger population to be maintained (Figure 8-3). It is important to realize that we have not incorporated stochasticity (natural variation) in the model, so it is possible that even though

harvest and operation of the dams is held constant, the population could decline if environmental variation was unfavorable to survival for several years in sequence. For a discussion of stochastic effects on simulation modeling, see the companion report by Emlen (1993).

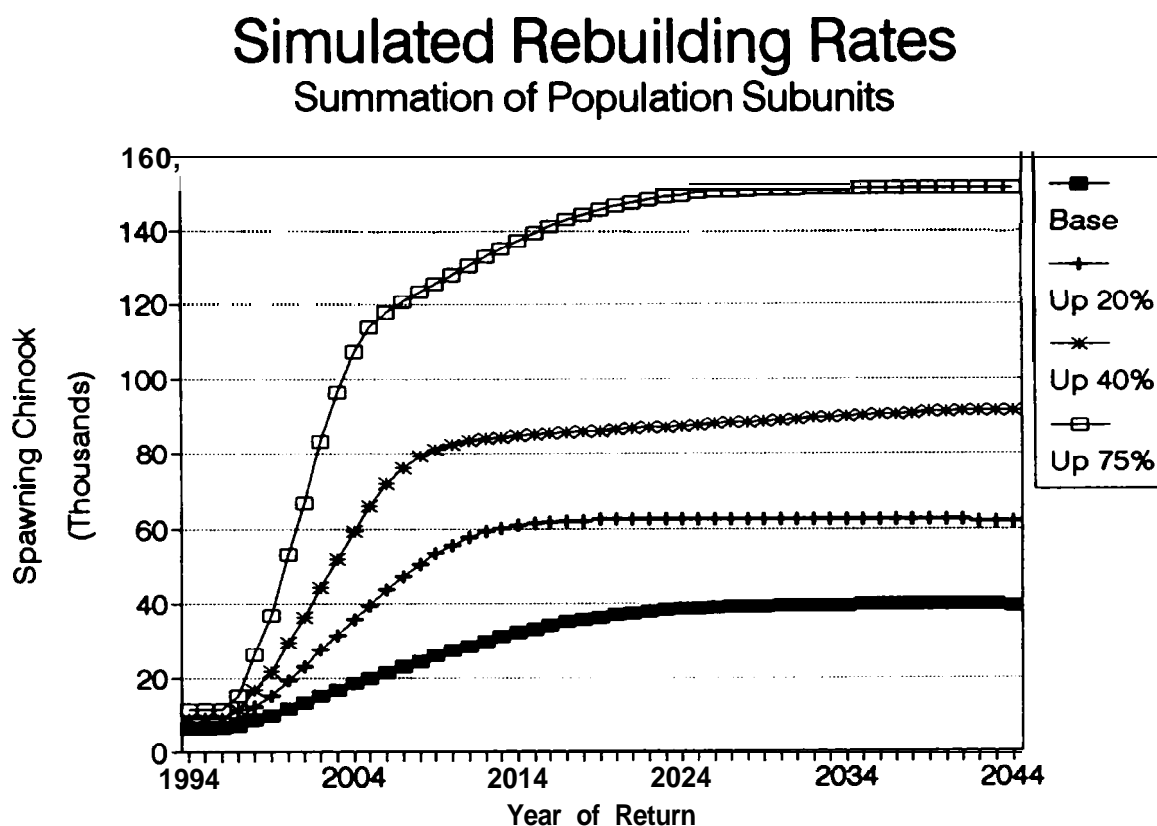


Figure 8-3. Simulated spawner abundance of spring chinook summed over the four habitat subunits during the next 50 years, given various increases in survival rate.

The most revealing result of this simulation is not how much the population increases, but what happens to the four subunits of the population as survival increases. Although a 20% increase in survival would cause about a 60% increase in the population above that for baseline conditions, the population subunit in poor quality habitat would still go extinct and the subunit in fair habitat would still decline (Figure 8-4). Only with a 40% increase in survival would the subunit in fair habitat begin to slowly increase, and with a 75% increase in survival, the fair subunit would increase for about 30 years when it would produce about five times as many adult fish as the excellent habitat (Figure 8-5). These simulations also demonstrate that the more productive subunits (those that produce more recruits per spawner) reach a new equilibrium sooner than less productive subunits.

These simulations illustrate the importance of accounting for differences in productivity of population subunits when de-listing criteria are set.

**RECOMMENDATION: We recommend that de-listing criteria be set to at least maintain a stable population in fair habitat and a growing population in good and excellent habitat for two generations.**

Such a criterion would guarantee population increases in good and excellent habitat, and would protect the subunit in fair habitat, which is about one third of all habitat available to spring/summer chinook in the basin. The subunit of the population rearing in poor habitat would be lost under this criteria, but if the habitat could be upgraded in the future, the area could be reseeded by chinook spawning and rearing in surrounding habitat. Espinosa and Lee (1991) demonstrated, as have others, that juvenile and adult chinook do locate and use enhanced habitat in stream sections of the Snake River basin where prior use had been at or near zero. We are not recommending here that a genetic subunit of the population be allowed to go extinct, but rather that lack of use of poor habitat should not be used as an indication of population endangerment.

Choice of a stable trend for chinook rearing in fair habitat as a de-listing criteria would buffer the population subunits in good and excellent habitat against the effects of a possible extended period of low ocean survival or drought. A frequency histogram of smolt-to-adult survival rates for wild spring chinook (basin average), as estimated by Raymond (1988), indicates that even after all mainstem dams were completed, survival rate varied 500% among broods (Figure 8-6). This variation, when compared to the limited range of survival rates between MSY and stock collapse (see Figure 5-3), necessitates that a survival rate for de-listing be chosen that would enable the subunits in good or excellent habitat to be maintained at levels near MSY.



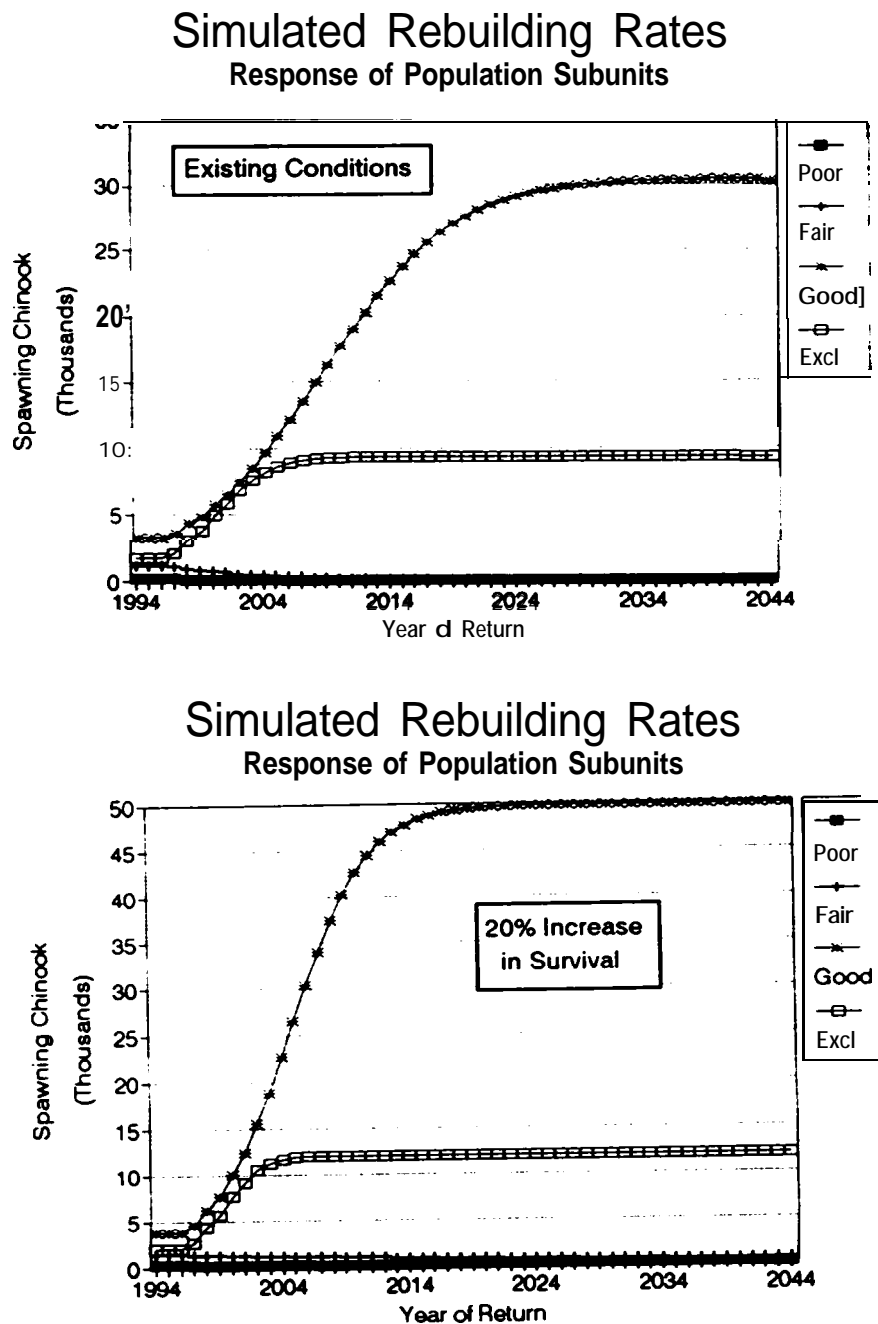


Figure 8-4. Simulated spawner abundance of spring chinook for each habitat subunit over the next 50 years, under existing conditions and with a 200% increase in survival.

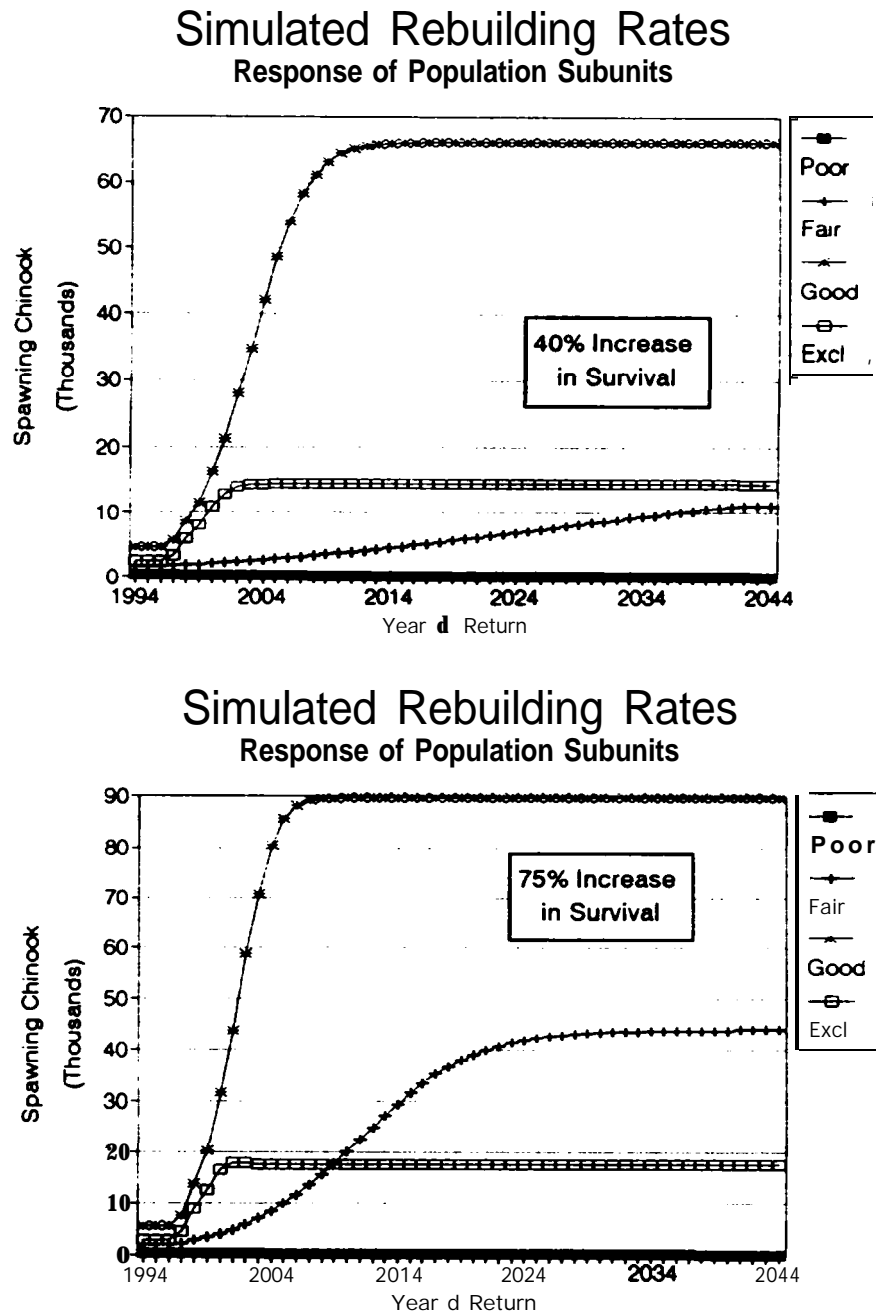


Figure 8-5. Simulated spawner abundance of spring chinook for each habitat subunit over the next 50 years, with a 40% and a 75% increase in survival.

## Smolt-to-Adult Survival Rates

### Snake River Spring/Summer Chinook

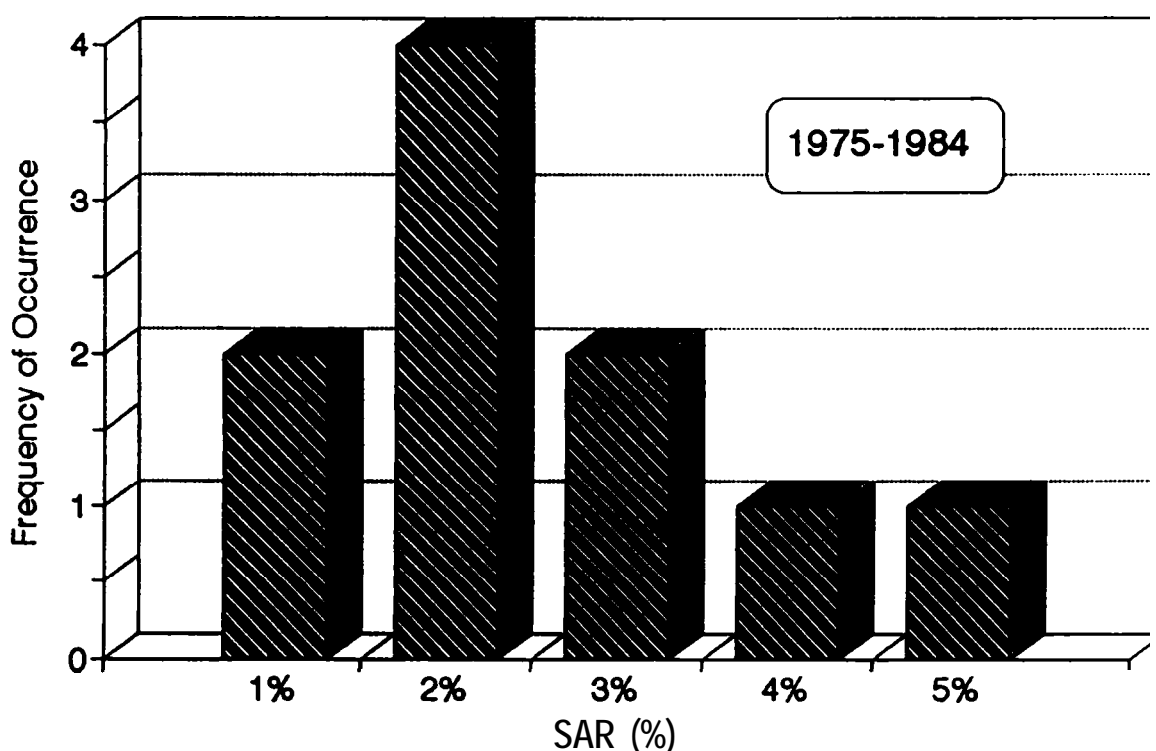


Figure 8-6. Frequency histogram of smolt-to-adult survival of Snake River spring chinook during 1964-1984 (adapted from Raymond 1988).

The simulation results indicate that in order to achieve the 70% of capacity criterion for recovery, an escapement of about 80,000 fish into good and excellent habitat would be required. We cannot claim to be accurate on this number, because the productivity parameter ( $a$ ) has only been guessed for different habitat qualities. The simulations indicate that somewhere between a 20% to 40% increase in survival rate would be necessary to maintain a stable population in fair habitat and reach 80,000 spawners in good and excellent habitat. Further, the simulations suggest that the escapement criteria could be achieved in about 15 years with a 40% increase in survival and about 10 years with a 75% increase in survival.

Our estimates of survival and the stock-recruitment parameters are imprecise and constitute critical uncertainties for determining stock recovery. Emphasis during the period of stock rebuilding must be placed on monitoring and the development of new information that will enable more accurate definition of when recovery has been achieved. McConnah and Anderson (1992) in their simulation analysis of rebuilding options for Snake River spring chinook came to a similar conclusion. Some of their conclusions bear repeating.

***"Major uncertainties exist which limit our ability to predict at this time the best course of action to achieve rebuilding." Further, "Because of this, rebuilding schedules, survival targets and the supporting analyses must be responsive to improvements in knowledge. . "It indicates the need for an adaptive and flexible approach guided by analysis of the existing information, and deliberate action to resolve pivotal uncertainties."***

The Snake River Salmon Recovery Team made similar recommendations in their draft Recovery Plan for Snake River Sockeye,

***"The Team recognizes and supports the need for building management flexibility that will adapt thereafter to changing conditions and will make full use of an evolving and expanding information base. .***

## 8.2 FALL CHINOOK

### 8.2.1 **Model Description**

We developed a population model for fall chinook that parallels the model just described for spring chinook. The model for fall chinook required its own unique set of parameter values, and it required additional accounting for age structuring of harvest and returns. The sequencing of life stages in the model is the same as that for spring chinook. Additional accounting for harvest was necessary, because fall chinook, unlike spring chinook, are highly vulnerable to the ocean fishery. The life cycle structure of the model is shown in Figure 8-7.

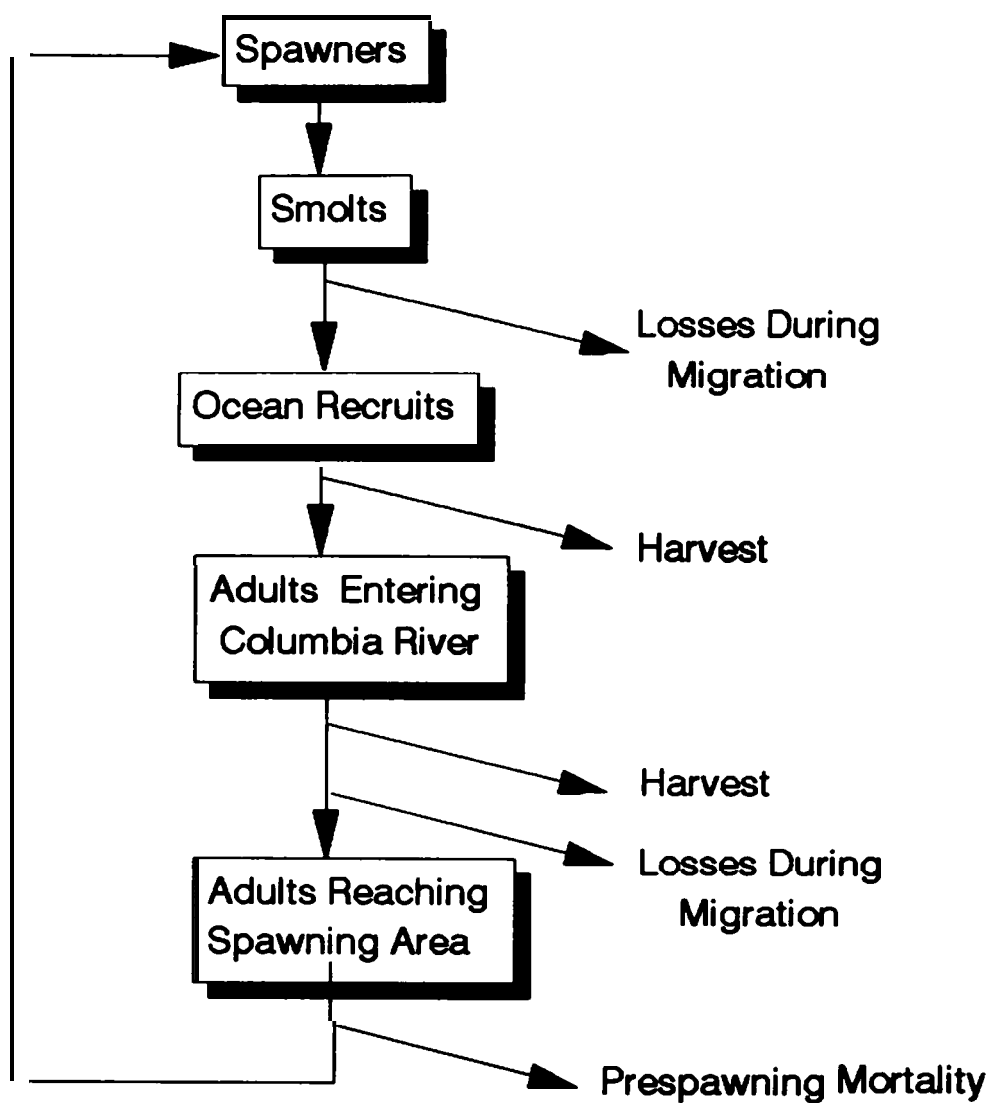


Figure 8-7. Flow diagram of our population model used to evaluate rebuilding schedules for fall chinook

The parameter values we used in the model for fall chinook are as follows:

Spawner Capacity	200,000
Ricker $\alpha$ (Adjusted to pre-dam conditions)	20
Ricker $\beta$ (spawner to spawner)	5E-06
Smolt-to-Adult survival rate (pre-dam)	12.6%
Smolt mortality per dam	22%
Ocean Harvest Rate	Annual values
Proportion 1-Salt	40%
Proportion 2-Salt	30%
Proportion 3-Salt	25%
Proportion 4-Salt	5%
River Harvest Rate	Annual values
Adult mortality per dam	5%
Prespawning Mortality Rate	10%

## 8.2.2 Simulation Results

### 8.2.1.1 Hindcasting

Initially, we simulated the population during 1965-1992 to evaluate how closely the simulated spawner abundances matched the observed abundance. The model was not intended to predict abundance, but rather to reflect trends in abundance, so that rebuilding rates could be estimated. Therefore, the intended comparison was between predicted and observed trends. We applied harvest rates in the ocean as estimated by PSC for the 1975-1983 broods of Upriver Bright fall chinook, and used the average rate of 45.8% for other years. We applied harvest rates in the river each year equal to those reported by WDF and ODFW (1992) for upriver fall chinook (Table 8-2). For this comparison, we assumed the spawner capacity for maximum production was 4,800 after 1966, as estimated by Schaller and Cooney (1992). Between 1961 and 1966 (Hells Canyon Dam was completed in 1967), we increased this number by an additional 6,600 spawners, which was the mitigation requirement for the river above Hells Canyon Dam, but below Oxbow Dam. The simulated returns match the observed declining trend in spawners closely (Figure 8-8), but the simulated population goes extinct near 1990. Thus, the population model we used for rebuilding analysis should be regarded as liberal in its allowances for mortality factors.

As for spring chinook the uncertainty in the stock-recruitment functions for fall chinook make it essential that the Recovery Plan emphasize a monitoring system that will enable feedback and correction of the stock-recruitment relationships as rebuilding progresses. The same types of monitoring will be required for fall chinook that were specified for spring chinook.

Table 8-2. Simulated spawner abundance and the annual harvest and survival rates used in the simulation model for fall chinook.

Spawning Year	Spawners	Adjusted SAR	Harvest Rates		Upstream Survival	Run Year	Adults Arrive Above Dams
			Ocean	River			
1930	266,702	12.6%	25.0%	71.9%	100%	1932	296,158
1931	266,542	12.6%	25.0%	71.9%	100%	1933	296,030
1932	266,427	12.6%	25.0%	71.9%	100%	1934	296,053
1933	266,448	12.6%	25.0%	71.9%	100%	1935	296,082
1934	266,474	12.6%	25.0%	71.9%	100%	1936	296,090
1935	266,481	12.6%	25.0%	69.5%	100%	1937	306,367
1936	275,730	12.6%	25.0%	73.5%	100%	1938	295,654
1937	266,088	9.8%	25.0%	64.1%	95%	1939	290,105
1938	261,095	9.8%	25.0%	72.7%	95%	1940	254,942
1939	229,448	9.8%	25.0%	72.3%	95%	1941	238,116
1940	214,304	9.8%	25.0%	68.6%	95%	1942	237,659
1941	213,893	9.8%	25.0%	80.2%	95%	1943	208,542
1942	187,688	9.8%	25.0%	70.5%	95%	1944	220,278
1943	198,250	9.8%	25.0%	69.7%	95%	1945	224,594
1944	202,134	9.8%	25.0%	80.1%	95%	1946	207,845
1945	187,060	9.8%	25.0%	76.4%	95%	1947	199,838
1946	179,854	9.8%	25.0%	75.0%	95%	1948	192,064
1947	172,858	9.8%	25.0%	67.0%	95%	1949	224,054
1948	201,648	9.8%	25.0%	75.8%	95%	1950	219,689
1949	197,720	9.8%	45.8%	40.9%	95%	1951	276,026
1950	248,423	9.8%	45.8%	74.1%	95%	1952	227,231
1951	204,508	9.8%	45.8%	67.3%	95%	1953	219,148
1952	197,233	7.7%	45.8%	67.4%	90%	1954	169,745
1953	152,771	7.7%	45.8%	67.1%	90%	1955	154,229
1954	138,806	7.7%	45.8%	48.6%	90%	1956	171,942
1955	154,748	7.7%	45.8%	25.1%	90%	1957	232,850
1956	209,585	6.0%	45.8%	29.3%	86%	1958	247,170
1957	222,453	6.0%	45.8%	56.7%	86%	1959	214,887
1958	193,398	6.0%	45.8%	46.2%	86%	1960	185,022
1959	166,520	6.0%	45.8%	53.9%	86%	1961	157,680
1960	141,912	6.0%	45.8%	44.1%	86%	1962	163,665
1961	147,299	4.7%	45.8%	47.1%	81%	1963	145,947

Table 6-2. Continued

Spawning Year	spawners	Adjusted SAR	Harvest Rates		upstream Run Survival	Year Arrive Above Dam	Adults
			Ocean	River			
1962	131,352	4.7%	45.896	57.7%	81%	1964	124,685
1963	112,216	5.2%	45.896	45.3%	81%	1965	118,665
1964	106,798	5.5%	45.8%	53.3%	01%	1966	113,240
1965	101,916	6.1%	46.896	38.096	81%	1967	135,586
1966	122,028	10.1%	45.0%	46.1%	81%	1966	164,019
1967	165,617	4.6%	45.896	52.7%	77%	1969	163,900
1968	147,510	5.3%	45.8%	50.8%	74%	1970	149,323
1959	134,391	4.4%	45.8%	59.7%	70%	1971	102,972
1970	92,674	3.1%	45.0%	54.9%	70%	1972	75,714
1971	68,143	1.6%	45.896	45.1%	70%	1973	48,993
1972	44,094	0.5%	45.8%	63.5%	70%	1974	25,233
1973	22,755	2.2%	45.8%	47.1%	70%	1975	15,703
1974	14,205	5.0%	45.8%	45.1%	66%	1976	14,840
1975	13,356	1.4%	45.8%	42.8%	66%	1977	12,137
1976	10,923	0.4%	45.8%	41.6%	66%	1978	6,100
1977	7,290	1.4%	45.8%	37.7%	66%	1979	4,191
1970	3,772	1.6%	52.0%	30.8%	66%	1980	2,432
1979	2,234	0.7%	60.0%	35.4%	66%	1981	1,707
1980	1,536	20%	45.0%	18.0%	66%	1982	1,397
1981	1,257	27%	28.0%	31.3%	66%	1983	1,346
1982	1,211	28%	38.0%	30.3%	66%	1984	1,535
1983	1,382	4.1%	55.0%	36.8%	66%	1985	1,647
1984	1,482	1.7%	39.0%	50.0%	66%	1986	1,304
1985	1,174	1.7%	37.0%	54.1%	66%	1987	985
1986	886	1.7%	420%	53.3%	66%	1988	645
1987	581	1.7%	45.8%	44.7%	66%	1989	451
1986	406	1.7%	45.8%	35.4%	66%	1990	337
1989	303	1.7%	45.896	30.4%	66%	1991	251



## Simulated Spawner Abundance

### Snake River Fall Chinook

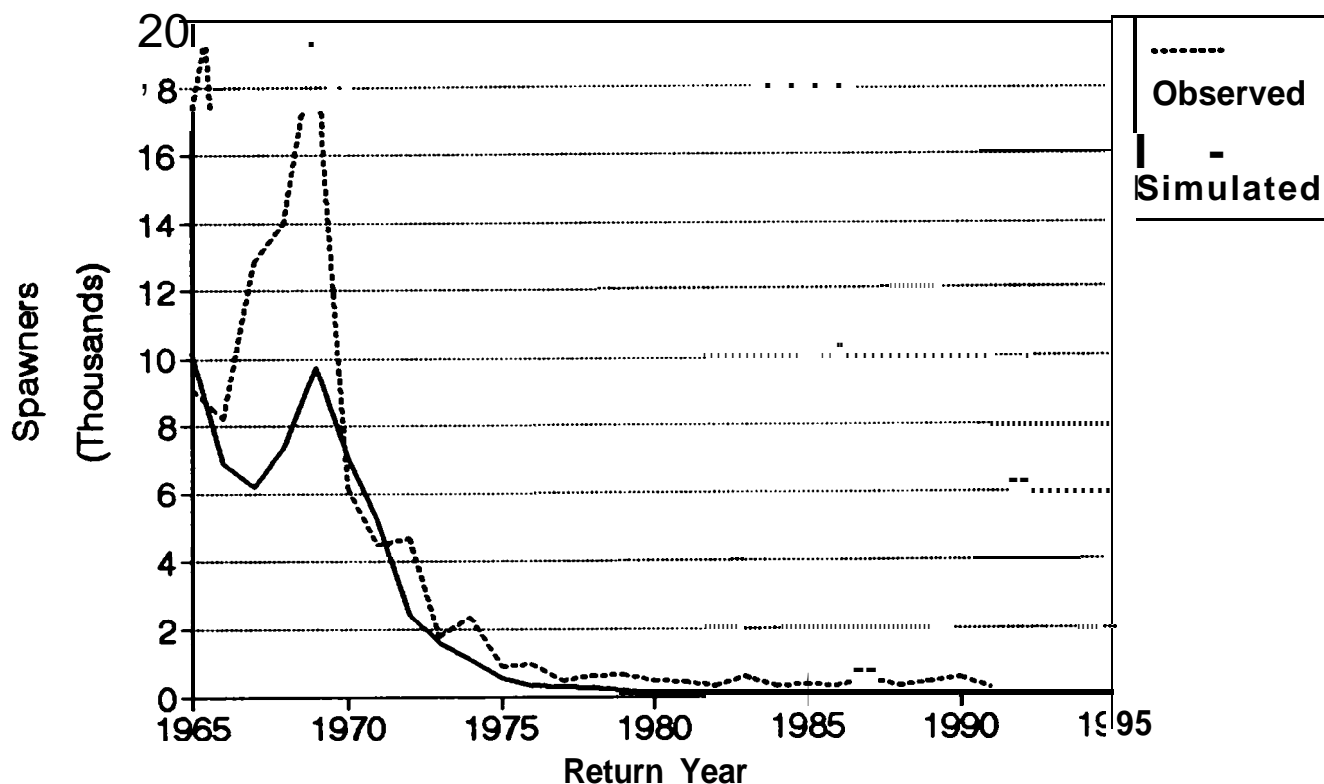


Figure 8-8. Comparison of simulated and actual returns of naturally produced fall chinook past the uppermost dam in the Snake River. Estimated actual returns from ODFW (1991). See Section 8 for a description of the simulation model.

**8.2.1.2 Forecasting**

We used the simulation model to forecast stock rebuilding over the next 50 years under six levels of increase in overall survival rates; 1) baseline conditions, 2) 10% increase in survival, 3) 20% increase in survival, 4) 40% increase in survival, 5) 60% increase in survival, and 6) 75% increase in survival. We also simulated the responses that would be achieved if rebuilding were promoted in the Snake River plus the Clearwater River and compared the response to that for the Snake River alone. An important consideration for recovery planning is that carrying capacity in the Snake River alone is estimated at 4,800 spawners, while in the Clearwater River it is estimated to be at least 191,000 spawners (Amsberg and Connor 1992). Fall chinook were eliminated from the Clearwater Basin in 1927 when their passage was blocked by Lewiston Dam, but Lewiston Dam was removed in 1973 and the basin is now fully accessible to fall chinook. We did not divide the population into habitat quality subunits as we did for spring chinook, because habitat quality and quantity have not been inventoried in the main stem Snake River, and habitat quality is likely to be less variable than for spring chinook.

**RECOMMENDATION: Survival rates of juvenile fall chinook during passage of mainstem dams should be monitored. Present estimates of juvenile passage mortality are only guesses. Simulated rates of stock recovery and the value of alternative management actions intended to produce recovery can also only be regarded as guesses without accurate estimates of downstream passage mortality.**

Increases in survival were expressed as a percentage of the existing survival, rather than as an absolute number of percentage points. Baseline survival rate was estimated to be 0.4% from homestream smolt to an ocean adult (12.6% pristine smolt-to-adult survival x 9.1% passage survival x 33.4% harvest survival). However, our hindcasting simulation resulted in extinction of the run by about 1990; whereas in actuality, the run has stabilized at 300 to 500 fish since 1960 (with the exception of 78 fish in 1990; a 1 in 10 year event). In order for the model to maintain a constant run of 400 fish, we had to double the base survival rate. Therefore, we used double the final survival rate in the hindcast simulations as the base rate in our simulations of rebuilding rates. The need to make this adjustment indicates that mortality factors were overestimated in our original assumptions. This is not surprising, given the limited data from which survival rates had to be derived. Harvest rates were held constant over the period of the simulations.

The simulations indicated that under existing conditions, the Snake River population

would remain constant at about 400 fish, but if habitat were seeded in the Clearwater Basin, the population would gradually increase to over 2,500 fish within 50 years (Figure 8-9). We again emphasize that we have not incorporated stochasticity (natural variation) into the model, so it could happen that if our parameter estimates are accurate, the population could decline due to environmental variation unfavorable to survival for several years in sequence. For a discussion of stochastic effects on simulation modeling, see the companion report by Emlen (1993).

Simulated spawner abundance for the next 50 years showed a notable increase in spawner abundance with a 20% increase in survival, and a dramatic increase up to about 80% of spawner capacity if survival increased 40% (Figure 8-10). The run size was predicted to be about 40 times greater if habitat in the Clearwater Basin was used. With existing habitat in the Snake River, the run size approached an equilibrium level within about 15 years of a 60% or 75% increase in survival (Figure 8-10).

### Simulated Fall Chinook Spawning Effect of Adding Clearwater River

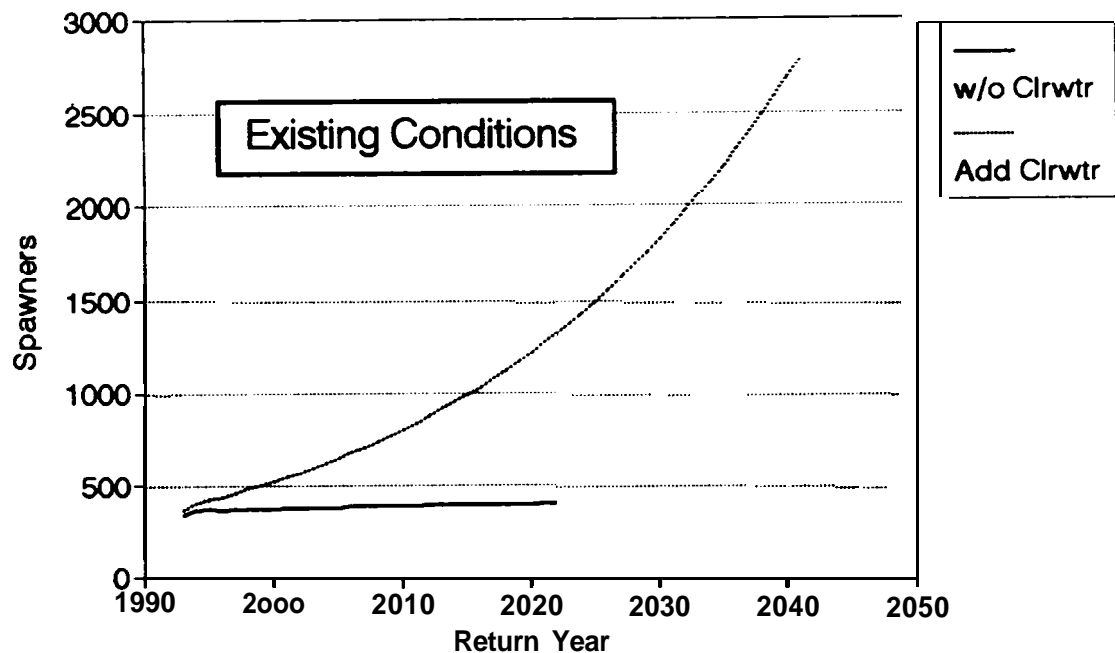


Figure 8-9. Comparison of simulated spawner abundance of fall chinook between habitat that includes the Clearwater Basin and habitat that excludes the Clearwater Basin, given that baseline conditions were maintained over the next 50 years.

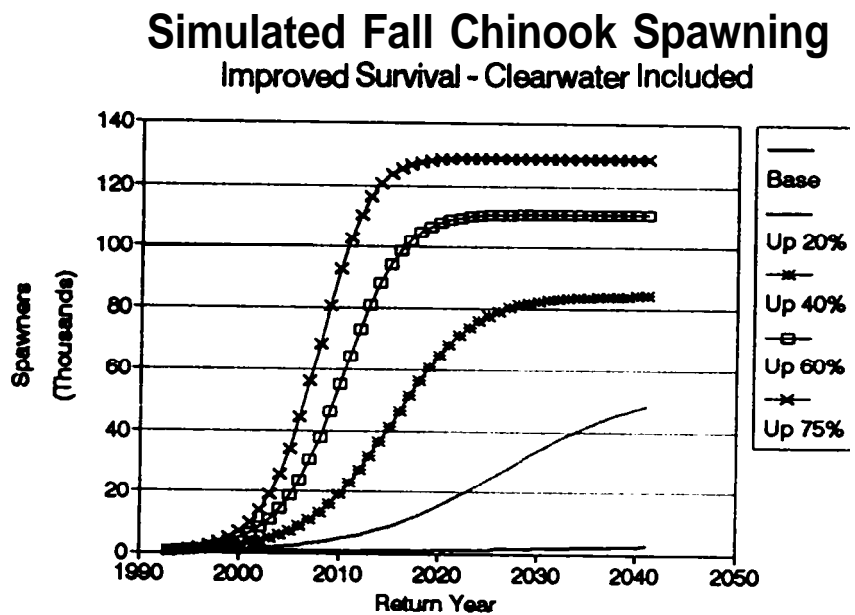
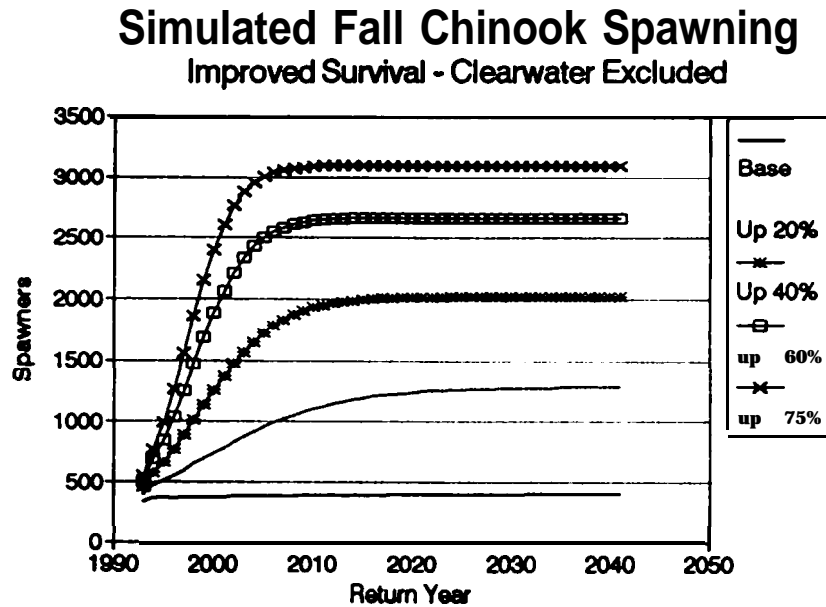


Figure 8-10. Simulated spawner abundance of fall chinook over the next 50 years, given increases in baseline survival of 20%, 40%, 60%, and 75%. Top graph excludes Clearwater Basin and bottom graph includes Clearwater Basin.

The simulation in which survival rate of fall chinook was increased 40% resulted in the population stabilizing after about 15 years at just above the spawner level required for 70% of MSY. Thus, a 30% to 40% increase in survival is the minimum that would achieve our de-listing criteria based on spawner abundance. Our criteria based on two generations with an increasing trend should be evaluated with a stochastic model, rather than the deterministic model developed here. Our simulations indicate that either a 60% or 75% increase in survival would enable the minimum spawner criterion to be achieved in two generations.

### **8.3 SOCKEYE**

Because recovery of the sockeye population is dependent on a captive brood program, it is not anticipated that significant numbers of adults will be available to spawn naturally for about two generations, or after 1998 (Bevan et al. 1992). The Snake River Salmon Recovery Team (Bevan et al. 1992) has developed projections of recovery rates that might be achieved by the recovery plan. The Team estimates that a 2.5 fold increase in smolt-to-adult survival rate, along with a doubling of the carrying capacity of Redfish Lake would be necessary to provide a self-sustaining population. Because of the dependence of the recovery on a captive brood program and the large uncertainty regarding the genetic viability of the population, we believe it is inappropriate to derive further conclusions from simulation modeling at this time. Therefore, we did no attempt to estimate rebuilding schedules. We refer the reader to the work of Bevan et al. (1992).

## **9. RECOMMENDATIONS**

### **De-Listing Criteria**

- **In order to be de-listed, the spring/summer and fall chinook populations should meet at least one of two criteria:**
  - 1) abundance of populations in good or excellent habitat should show a significant exponential increase for at least two generations. Simulations indicate that a minimum of two generations are required for the population to begin to stabilize. Populations in fair habitat should be stable for at least two generations.**
  - 2) spawner abundance or parr density should average 70% of carrying capacity for one generation.**
- **We recommend that redd counts and parr densities be the primary measures of abundance and trend. This may not be possible for all listed stocks.**
- **Recovery should be evaluated separately for summer and spring components within each subbasin and criteria should be met for each component within each subbasin.**
- **In assessing the recovery program, we recommend that redd counts not be expanded by spawner number, and that log ratios between unexpanded return and brood-year redds be used to assess trend.**
- **De-listing criteria for Redfish Lake sockeye should be developed from information gathered on stock productivity as the stock rebuilds. A thorough monitoring program for smolt abundance and adult recruits should form the basis for characterizing the stock-recruitment relationship of Redfish Lake sockeye. That relationship should be examined to determine if there is a mortality level beyond which the population declines rapidly, in a similar manner to what has been estimated for spring and fall chinook. If such a mortality level exists, it should be used in some form in the de-listing criteria.**

**Analytical Methods****Extinction Models**

- ▶ **Any model used to assess extinction probabilities should accommodate both variation in the parameter estimate and random variation. The model should be checked for its sensitivity to extinction number and should use lag-times that reflect the generation length.**
- ▶ **The extinction number should correspond to the point at which all fish would be taken into a captive breeding program to prevent complete extirpation of the stock.**
- ▶ **For spring/summer chinook, variation in the parameter estimate can be characterized by estimating separate trend parameters from comparable index areas and using them to estimate the variance in parameter estimates.**

**Trend Analysis**

- ▶ **The various methods of estimating the exponential rate of growth for the recovery period and of decay for the eight-dam historic era should be investigated and used to evaluate population growth during early part of the recovery program. The exponential model should be compared to other growth models to confirm that growth is exponential in nature.**
- ▶ **Since the  $\beta$  parameter of the Ricker function is not precisely measured during the exponential phase of growth, the Ricker function should be evaluated in later stages of the recovery program. At that time, the population should be tested as to whether its point of spawners for maximum production,  $1/\beta$ , is characteristic of a healthy population.**
- ▶ **Use the whole of the pre-recovery eight-dam era as a base and adjust the abundance for a drought covariate and possibly other environmental covariates.**
- ▶ **Trend and abundance measures in the recovery period should be also be adjusted for environmental factors.**

- Unless sources of bias can be measured and adjusted for, the following should **not** be used for assessing recovery:
  - Abundance or trend estimates based on reconstructed counts,
  - Extinction model parameter estimates based on reconstructed counts,
  - Historical and current redd counts as a measure of absolute abundance.
- Based on our investigations to date, trends will be more powerfully assessed using means of trends over index areas rather than fitting a trend to the pooled count. Therefore, we recommend that historical trends be separately estimated for each index area and appropriately averaged for the purpose of statistical tests. This does not preclude trend analysis on pooled counts. The level of trend summarization, whether based on means or pooled totals, should be based on such demographic characteristics as those discussed in this report.
- The recovery program should try to ascertain whether the decline in the summer proportion of the spring/summer dam counts is due to human activities. It should take action to guarantee that the summer portion of the run does not decline further, and it should consider possible actions that would lead to the reestablishment of the historic run distribution.

### **Monitoring**

- The use of the clipped adipose fin as a mark should be desequestered from the coded wire tag so that the mark can be used to differentiate hatchery fish from the wild/natural listed species.

### **Spring/Summer Chinook**

- Stratified random sampling should be used in selecting new redd count areas; current index areas should continue to be assessed for the foreseeable future.



- ▶ The same age classification procedures should be followed over agencies. Specifically, we recommend that the actual age, not the fork length, of the fish be tallied. Age classifications should be reported separately for each sex.
- ▶ All dead fish on spawning surveys are currently counted, whether or not they are spawned out. A separate tally should be given for hatchery and natural spawned-out carcasses.
- ▶ The number of redds in an index area should be multiplied by the wild/natural proportion of spawned-out female carcasses to estimate the number of wild-origin redds. This estimate should then be the primary focus of abundance, trend and extinction assessments. Assessments based on total redds within enumerated areas would still continue because the pre-recovery reference base would have been based on total redds per index area.
- ▶ Develop integrated sampling strategies for parr density and redd count monitoring to permit the recovery target values based on redd counts to take seeding capacity into account.

#### Fall **Chinook**

- ▶ Given the uncertainty in stock-recruitment functions, it is essential that the Recovery Plan emphasize a monitoring system that will enable feedback and correction of the stock-recruitment relationships as rebuilding progresses. This will require 1) estimates of successful spawner abundance, 2) estimates of smolt survival past all eight dams, 3) estimates of harvest, and 4) estimates of adult losses at dams.
- ▶ Survival rates of juvenile fall chinook during passage of mainstem dams should be monitored. Present estimates of juvenile passage mortality are only guesses. Simulated rates of stock recovery and the value of alternative management actions intended to produce recovery should also only be regarded as guesses without accurate estimates of downstream passage mortality.

- **Correct historic reconstructed dam count bias associated with night passage by making adjustments using 24-hour counts from the 1992 field test data for video-imagery conducted by the Columbia River Inter-tribal Fish Commission. The bias assessment should be continued until the variability in night passage has been well established and an adjustment factor with acceptable accuracy and precision can be estimated.**
- **We recommend using the 24-hour video counts in place of viewing-window counts if each fish is clearly visible in its entirety on at least one frame to permit length and mark assessment.**
- **Annual indices of juvenile fall chinook abundance leaving the Snake River should be continued. Work should continue to resolve uncertainty in race identification and in trapping efficiency. The relationship between spawner abundance and smolt abundance, as well as factors influencing the relationship, should be established.**
- **In the future, if Lyons Ferry Hatchery takes natural fish as brood stock, the age and sex distributions should be tallied separately for the natural and hatchery fish. However, the number of natural fish taken may be too small for any precise assessment as to the actual distribution of these demographic characteristics.**
- **Information on passage survival is critical to understanding of the population dynamics of fall chinook. Procedures should be developed and implemented to estimate mortality of juvenile fall chinook during their outmigration through the Snake and Columbia Rivers.**
- **Smolt-to-adult survival rate of fall chinook should be estimated directly from estimates of Snake River smolt abundance and adult recruits.**

### **Sockeye**

- **A thorough monitoring program for smolt abundance and adult recruits should form the basis for characterizing the stock-recruitment relationship of Redfish Lake sockeye as the stock rebuilds. That relationship should be examined to determine if there is a mortality level beyond which the population declines rapidly, in a similar manner to what has been estimated for spring/summer and fall chinook. If such a mortality level exists, it should be used in some form in the de-listing criteria.**

7. The Recovery Plan should include development of a method to distinguish ocean adult returns from those that mature in reservoirs downstream of Redfish Lake. It may be necessary to sample scales from returning adults and subject the scales to strontium analysis to distinguish ocean migrants.

## **REFERENCES**

- Allendorf, F.W. 1991. Evaluation of 'Determining minimum viable populations under the endangered species act' by G. G. Thompson (NOAA Tech. Memo. NMFS F/NWC-198). Report to U. S. Department of Energy, Bonneville Power Administration, Contract DE-AP79-91BP21862. Portland, Oregon.
- Allendorf, F.W. 1991. Evaluation of genetic issues for Snake River fall chinook salmon. Report to U. S. Department of Energy, Bonneville Power Administration, Contract DE-AP79-91BP21862. Portland, Oregon.
- Allendorf, F.W. 1991. Evaluation of genetic issues for Snake River spring and summer chinook salmon. Report to U. S. Department of Energy, Bonneville Power Administration, Contract DE-AP79-91BP21862. Portland, Oregon.
- Arnsberg, B.D., and E. Connor. 1992. Mainstem Clearwater River study: Assessment for salmonid spawning, incubation, and rearing. Report to Bonneville Power Administration, Contract DE-BI79-87BP37474. Portland, Oregon.
- Avery, D.J. 1992. Memo: IDFG juvenile and adult trapping program, June 1, 1992.
- Ayerst, J.D. 1977. The role of hatcheries in rebuilding steelhead runs of the Columbia River system. p. 84-88 In E. Schweibert [ed] Columbia River salmon and steelhead. Am. Fish. Soc. Spec. Publ. No. 10.
- Bell, M.C., Z.E. Parkhurst, R.G. Porter, and M. Stevens. 1976. Effects of power peaking on survival of juvenile fish at lower Columbia and Snake River dams. U.S. Army Corps of Engineers. (DACW57-75-C-0173). Portland, Oregon.
- Bennett, D.E. and P. Hirose. 1972. Northeastern Oregon spring chinook spawning ground surveys for 1972. Fish Commission of Oregon. Clackamas, Oregon.
- Bennett, D.E. 1973. Northeastern Oregon spring chinook spawning ground surveys for 1973. Fish Commission of Oregon. Clackamas, Oregon.
- Bennett, D.E. 1975. Northeastern Oregon spring chinook spawning ground surveys for 1974. Fish Commission of Oregon. Clackamas, Oregon.

- Bennett, D.E. 1975. Northeastern Oregon spring chinook spawning ground surveys for 1975. Fish Commission of Oregon. Clackamas, Oregon.
- Berkson, J. 1991. Lyons Ferry fall chinook coded wire tag analysis. Columbia River Inter-tribal Fish Commission, File report, October 30, 1991. Portland, Oregon.
- Bevan, D. E., J. Harville, P. Bergman, T. Bjornn, J. Crutchfield, P. Klingeman, and J. Litchfield. 1992. Initial measures necessary for recovery of Snake River sockeye salmon. Draft for informal review, not yet submitted to the National Marine Fishery Service (Pre-decisional ESA document). September 24, 1992. Snake River Salmon Recovery Team.
- Bjornn, T.C., D. Craddock, and D. Corley. 1968. Migration and survival of Redfish Lake, Idaho, sockeye salmon Oncorhynchus nerka. American Fisheries Society, 97 (4): 360-373.
- Bjornn, T.C., R.R. Ringe, K.R. Tolotti, P.J. Keniry, and J.P. Hunt. 1992. Migration of adult chinook salmon and steelhead past dams and through reservoirs in the lower Snake River and into tributaries - 1991. Idaho Cooperative Fish and Wildlife Research Unit, University of Idaho, Moscow, Idaho. Annual Report for 1991. Technical Report 92-2, U. S. Army Corps of Engineers, Walla Walla, Washington.
- Bjornn, T.C. and C.A. Peery. 1992. A review of literature related to movements of adult salmon and steelhead past dams and through reservoirs in the lower Snake River. Technical Report 92-1, U.S. Army Corps of Engineers, Walla Walla, Washington.
- Bohn, B. 1967. Northeastern Oregon spring chinook spawning ground surveys for 1967. Fish Commission of Oregon. Clackamas, Oregon.
- Bowles, E. and T. Cochnauer. 1984. Potential sockeye salmon production in Alturas Lake Creek drainage, Idaho. Idaho Department of Fish and Game, P.O. Number 40, 0267-4-127.373. Boise, Idaho.
- Bonneville Power Administration. 1984. 1984 Annual Report from the Water Budget Managers to the Northwest Power Planning Council. Portland, Oregon.

- Brege, D.A., W.T. Norman, G.A. Swan, and J.G. Williams. 1988. Research at McNary Dam to improve fish guiding efficiency of yearling and subyearling chinook salmon, 1987. CZES, NMFS, report to U.S. Army Corps of Engineers, Contr. No. DACW88-84-H-0034. Portland, Oregon.
- Broderick, S. 1990. Notice of petition - To Secretary of the Interior. RE: Listing Snake River race sockeye salmon as endangered. Shoshone-Bannock Tribes, Fort Hall, Idaho.
- Busack, C. 1991. Genetic evaluation of the Lyons Ferry Hatchery stock and wild Snake River fall chinook. Report submitted to the ESA Administration Record for fall chinook salmon, May 1991, 59p. Washington Department of Fisheries. Olympia, Washington.
- Ceballos, J.R., S.W. Pettit, and J.L. McKern. 1992. Transport operations on the Snake and Columbia Rivers. Fish Transportation Oversight Team (FTOT) Annual Report- FY 1991, NOAA Tech. Memo Nmfs F/NWR-18. Seattle, Washington.
- Chapman, D.W., J. Van Hying, and D. McKenzie. 1982. Alternative approaches to base run and compensation goals for Columbia River salmon and steelhead resources. Battelle Pacific Northwest Laboratories, Contract 231 1204050, Richland, Washington.
- Chapman, D.W., and K.L. Witty. 1993. Habitats of weak salmon stocks of the Snake River basin and feasible recovery measures. Report submitted to Bonneville Power Administration. Portland, Oregon.
- Chapman, D.W. 1988. Critical review of variables used to define effects of fines in redds of large salmonids. Trans. Am. Fish. Soc. 117:1-21.
- Chapman, D.W., W.S. Platts, D. Park, and M. Hill. 1990. Status of Snake River sockeye salmon. Submitted to Pacific Northwest Utilities Conference Committee. Portland, Oregon.
- Chapman, D., A. Giorgi, M. Hill, A. Maule, S. McCutcheon, D. Park, W. Platts, K. Pratt, J. Seeb, L. Seeb, and F. Utter. 1991. Status of Snake River chinook salmon. Report submitted to Pacific Northwest Utilities Conference Committee, Portland, Oregon.

- Columbia Basin Fish and Wildlife Authority. 1991. Integrated system plan for salmon and steelhead production in the Columbia River basin. Columbia Basin System Planning. 91-16. Portland, Oregon.
- Connor, W.P., H.L. Burge, and W.H. Miller. 1992. Identification of the spawning, rearing, and migratory requirements of fall chinook salmon in the Columbia River Basin. In: D.W. Rondorf and W.H. Miller (ed.). Submitted to Lower Snake River Compensation Plan Office and Bonneville Power Administration. Contract DE-AI79-91 BP21 708. Portland, Oregon.
- Demory, R. 1964. Northeastern Oregon spring chinook spawning ground surveys for 1964. Fish Commission of Oregon. Clackamas, Oregon.
- Demory, R. 1965. Northeastern Oregon spring chinook spawning ground surveys for 1965. Fish Commission of Oregon. Clackamas, Oregon.
- Dennis, B. and M. Taper. 1993 draft. Density dependence in time series observations of natural populations: estimations and testing. Manuscript submitted to: Ecology.
- Dennis, B., P. L. Munholland, and J. M. Scott. 1991. Estimation of growth and extinction parameters for endangered species. Ecological Monographs, 61(2): 115-143.
- Donaldson, L.R. 1970. Selective breeding salmonid fishes, pages 65-74. In: W.J. McNeil (ed.). Marine Aquaculture. Oregon State University Press, Corvallis, Oregon.
- Downey, T.W., G.L. Susac, and J.W. Nicholas. 1986. Research and development of Oregon's coastal chinook salmon stocks. Oregon Department of Fish and Wildlife, Annual Report, Fish Research Project NA-85-ABD-00115. Portland, Oregon.
- Evermann, B.W. 1897. Salmon investigations in the headwaters of the Columbia River in the State of Idaho, in 1895, together with notes upon the fishes observed in that state in 1894 and 1895. Report. U.S. Fish Commission Bulletin 16: 149-202.
- Falconer, D.S. 1981. Introduction to quantitative genetics. Longman, Inc., New York, N.Y. 340p.

- Fast, D., J. Hubble, M. Kohn, and B. Watson. 1991. Yakima River spring chinook enhancement study. Yakima Indian Nation Fisheries Resource Management. Report submitted to Bonneville Power Administration, Project 82-16, 345p. Portland, Oregon.
- Fish Passage Center. 1984. Fish passage managers 1989 annual report. Columbia Basin Fish and Wildlife Authority. Portland, Oregon.
- Fish Passage Center. 1985. Fish passage managers 1989 annual report. Columbia Basin Fish and Wildlife Authority. Portland, Oregon.
- Fish Passage Center. 1986. Fish passage managers 1989 annual report. Columbia Basin Fish and Wildlife Authority. Portland, Oregon.
- Fish Passage Center. 1987. Fish passage managers 1989 annual report. Columbia Basin Fish and Wildlife Authority. Portland, Oregon.
- Fish Passage Center. 1987. Migrational characteristics of Columbia basin salmon and steelhead trout, 1986. Smolt Monitoring Program Annual Report 1986, Vol. I. Contract DE-AI79-86BP61747. Portland, Oregon.
- Fish Passage Center. 1988. Fish passage center 1990 annual report. Columbia Basin Fish and Wildlife Authority. Portland, Oregon.
- Fish Passage Center. 1989. Fish passage managers 1989 annual report. Columbia Basin Fish and Wildlife Authority. Portland, Oregon.
- Fish Passage Center. 1990. Fish passage managers 1989 annual report. Columbia Basin Fish and Wildlife Authority. Portland, Oregon.
- Fish Passage Center. 1991. Fish passage center 1990 annual report. Columbia Basin Fish and Wildlife Authority. Portland, Oregon.
- Fish Passage Center. 1992. Fish passage center 1991 annual report. Columbia Basin Fish and Wildlife Authority. Portland, Oregon.



- Fisher, T.R. 1992. Input parameters for the modeling of Snake River salmon with the Columbia River salmon passage model (CRiSP.0). In: U.S. Army Corps of Engineers. 1992. Interim Columbia and Snake Riverflow improvement measures for salmon, draft supplemental environmental impact statement (SEIS). Walla Walla, Washington.
- Gall, G.A., J. Baltodano, and N. Huang. 1988. Heritability of age at spawning for rainbow trout. *Aquaculture* 68:93-102.
- Garrison, R.C., and M.M. Rosentreter. 1981. Stock assessment and genetic studies of anadromous salmonids. Federal Aid Progress Rpts. Fisheries, 1980. Oregon Dept. Fish and Wildlife, Fish Div. 66p. Portland, Oregon.
- Gharrett, A.J., and S.M. Shirley. 1985. A genetic examination of spawning methodology in a salmon hatchery. *Aquaculture* 47:245-256.
- Hager, R. C., and C.W. Hopley. 1981. A comparison of the effect of adult return timing of Cowlitz and Toutle hatchery coho on catch and escapement. Washington Department of Fisheries, Technical Report No. 58. Olympia, Washington.
- Hall-Griswold, J.A. 1990. Sockeye of Stanley Basin summary. Idaho Department of Fish and Game. Boise, Idaho.
- Hankin, D.G. 1990. Effects of month of release of hatchery-reared chinook salmon on size at age, maturation schedule, and fishery contribution. Oregon Department of Fish and Wildlife, Information Reports (Fish) 90-4, Portland, Oregon.
- Healey, M.C. 1982. Catch, escapement, and stock-recruitment for British Columbia chinook salmon since 1951. Canadian Tech. Report of Fish. and Aquatic Sci. 1107.
- Hirose, P. 1971. Northeastern Oregon spring chinook spawning ground surveys for 1971. Fish Commission of Oregon. Clackamas, Oregon.
- Hirose, P. 1978. Northeastern Oregon spring chinook and summer steelhead spawning ground surveys for 1977. Oregon Department of Fish and Wildlife. Portland, Oregon.

- Hirose, P. 1979. Northeastern Oregon spring chinook and summer steelhead spawning ground surveys for 1978. Oregon Department of Fish and Wildlife. Portland, Oregon.
- Hirose, P. 1980. Northeastern Oregon spring chinook and summer steelhead spawning ground surveys for 1979. Oregon Department of Fish and Wildlife. Portland, Oregon.
- Hirose, P. 1982. Northeastern Oregon spring chinook and summer steelhead spawning ground surveys for 1967-81. Oregon Department of Fish and Wildlife Portland, Oregon.
- Hirose, P. 1984. Northeastern Oregon spring chinook and summer steelhead spawning ground surveys for 1967-83. Oregon Department of Fish and Wildlife. Portland, Oregon.
- Hirose, P. 1984. Northeastern Oregon spring chinook and summer steelhead spawning ground surveys for 1967-84. Oregon Department of Fish and Wildlife. Portland, Oregon.
- Hillman, T.W. and D.W. Chapman. 1989. Abundance, growth, and movement of juvenile chinook salmon and steelhead. Summer and winter ecology of juvenile chinook salmon and steelhead trout in the Wenatchee River, Washington. Report submitted to Chelan County Public Utility District Wenatchee, Washington.
- Homer, N., and T.C. Bjornn. 1981. Status of upper Columbia and Snake River spring chinook salmon in relation to the Endangered Species Act. Univ. Idaho Coop. fish. Res. Unit. Moscow, Idaho. Rep. U.S. Fish Wildl. Serv.
- Idaho Department of Fish and Game. 1978. 1978 Salmon Spawning Ground Surveys. Boise, Idaho.
- Idaho Department of Fish and Game. 1980. 1979 Salmon Spawning Ground Surveys. Boise, Idaho.
- Idaho Department of Fish and Game. 1982. 1981 Salmon Spawning Ground Surveys. Boise, Idaho.

- Idaho Department of Fish and Game. 1983. 1982 Salmon Spawning Ground Surveys. Boise, Idaho.
- Idaho Department of Fish and Game. 1984. 1983 Salmon Spawning Ground Surveys. Boise, Idaho.
- Idaho Department of Fish and Game. 1985. 1984 Salmon Spawning Ground Surveys. Boise, Idaho.
- Idaho Department of Fish and Game. 1986. 1985 Salmon Spawning Ground Surveys. Boise, Idaho.
- Idaho Department of Fish and Game. 1988. 1986 Salmon Spawning Ground Surveys. Boise, Idaho.
- Idaho Department of Fish and Game. 1988. 1987 Salmon Spawning Ground Surveys. Boise, Idaho.
- Idaho Department of Fish and Game. 1990. 1988 Salmon Spawning Ground Surveys. Boise, Idaho.
- Idaho Department of Fish and Game. 1992. Anadromous fish management plan 1992-1996. Boise, Idaho.
- Jacobs, S.E. and C.X. Cooney. 1991. Improvements of methods used to estimate the spawning escapement of Oregon coastal natural coho salmon. Oregon Department of Fish and Wildlife, Portland, Oregon.
- Johnson, D.R., W.M. Chapman, and R.W. Schoning. 1948. The effects of salmon populations of the partial elimination of fixed fishing gear on the Columbia River in 1935. Oregon Fish Commission. Contribution No. 11. Portland, Oregon.
- Jonasson, B.C., R.W. Carmichael, and R.T. Messmer. 1992. Estimating natural escapement of spring chinook salmon: An evaluation of index spawning ground surveys in Northeast Oregon. Paper contributed to the Idaho American Fish Society meetings, March 1992, Boise, Idaho.

- Koski, C.H., S.W. Pettit, J.B. Atheam, and A.L. Heindl. 1987. Transport operations on the Snake and Columbia Rivers. Fish Transportation Oversight Team (FTOT) Annual Report-FY 1986, NOAA Tech. Memo NMFS F/NWR-18. Seattle, Washington.
- Lee, D.C. and J.B. Hyman. 1991. The stochastic life-cycle model (SLCM): a tool for simulating the population dynamics of anadromous salmonids. Resources for the Future. Washington, D.C.
- Lestelle, L.C. and L.G. Gilbertson. 1993. Harvest management and recovery of Snake River salmon stocks. Mobrand Biometrics, Inc. Report submitted to Bonneville Power Administration. Portland, Oregon.
- Lindsay, R.B., B.C. Jonasson, and R.K. Schroeder. 1989. Spring chinook salmon in the Deschutes River, Oregon. Oregon Department of Fish and Wildlife, Information Report 89-4. Portland, Oregon.
- Liscom, K. L. and L.C. Stuehrenberg. 1983. Radio tracking studies of upriver bright fall chinook salmon between Bonneville and McNary dams, 1982. National Marine Fisheries Service, Northwest and Alaska Fisheries Center, Final Report, Seattle, Washington.
- Lister, D.B., and H. Genoe. 1970. Stream habitat utilization by cohabiting underyearlings of chinook (*Oncorhynchus tshawytscha*) and coho (*O. kisutch*) salmon in the Big Qualicum River, British Columbia. J. Fish. Res. Bd. Canada 27:1215-1224.
- Major, R.L. and D.R. Craddock. 1962. Influence of early maturing females on reproductive potential of Columbia River blueback salmon (*Oncorhynchus nerka*). Fish and Wildlife Service Fishery Bulletin 194, Volume 61.
- Matthews, G.M. and R.S. Waples. 1991. Status review report Snake River spring/summer chinook salmon. NOAA Technical Memorandum NMFS F/NWC-200. U. S. Department of Commerce, National Oceanic and Atmospheric Administration, Seattle, Washington.
- McKenzie, D., D. Weitkamp, T. Schadt, D. Carlile and D. Chapman. 1983. 1982 Systems mortality study. 27 p. + appendices.

- McKenzie, D., D. Carlile, and D. Weitkamp. 1984. 1983 Systems mortality study. 25 p. + appendices.
- Mendel, G., D. Milks, R. Bugert, and K. Petersen. 1992. Upstream passage and spawning of fall chinook salmon in the Snake River, 1991. Washington Department of Fisheries. Report submitted to U.S. Fish and Wildlife Service. Agreement 14-16-0001-91502. Boise, Idaho.
- Mullan, J.W. 1986. Determinants of sockeye salmon abundance in the Columbia River, 1800's-1982: A review and syntheses. U.S. Department of the Interior, U.S. Fish and Wildlife Service, Biological Report 86 (12).
- Mullan, J.W. 1990. Status of chinook salmon stocks in the mid-Columbia. pp. 45-56. IN: D. Park, Convenor, Status and future of spring chinook salmon in the Columbia River basin – conservation and enhancement. Spring chinook Salmon Workshop, 8-9 November, 1989, Pasco Washington, sponsored by the National Marine Fisheries Service. 130 pp.
- National Marine Fisheries Service. 1991. Factors for decline: A supplement to the notice of determination for Snake River fall chinook salmon under the endangered species act. National Marine Fisheries Service, Environmental and Technical Services Division, Portland, Oregon.
- National Marine Fisheries Service. 1992. 1992 Operation of the federal Columbia River power system. Endangered Species Act, Section 7 consultation/conference, biological opinion. National Marine Fisheries Service, Northwest Region. Portland, Oregon.
- Nei, M. 1978. Estimation of average heterozygosity and genetic distance from a small number of individuals. Genetics 89:583-590.
- Nicholas, J.W., T.W. Downey, D. Bottom, and A. McGie. 1984. Research and development of Oregon's coastal chinook stocks. Oregon Department of Fish and Wildlife, Fish Research Project 82-ABD-ORIE, Annual Report, Portland, Oregon.

- Nicholas, J.W. and D.G. Hankin. 1988. Chinook salmon populations in Oregon coastal river basins: description of life histories and assessment of recent trends in run strengths. Oregon Department of Fish and Wildlife, Information Reports (Fish) 88-1. Northwest Power Planning Council (NPPC). 1989. Salmon and steelhead system planning documentation. Prepared by the Monitoring and Evaluation Group. Portland, Oregon.**
- North Pacific Division Corps of Engineers. 1991. Annual fish passage report: Columbia and Snake Rivers salmon, steelhead and shad. Prepared by: U. S. Army Engineer Districts, Portland, Oregon and Walla Walla, Washington.**
- Oregon Department of Fish and Wildlife and four other Agencies. 1989. Grande Ronde River subbasin: Salmon and steelhead production plan. Columbia Basin System Planning. Portland, Oregon.**
- Oregon Department of Fish and Wildlife and Washington Department of Fisheries. 1990. Status report. Columbia River fish runs & fisheries, 1938-1989. Portland, Oregon.**
- Oregon Department of Fish and Wildlife 1991. Snake River chinook salmon: Evaluation of stock status, causes of decline and critical habitat. Report submitted to National Marine Fisheries Service, Seattle, Washington.**
- Oregon Department of Fish and Wildlife and Washington Department of Fisheries. 1991. Report concerning commercial seasons for spring chinook, sturgeon, shad, and other fisheries and miscellaneous regulations for 1991. January 18, 1991. 49 p. Portland, Oregon.**
- PSC (Pacific Salmon Commission). 1990. Chinook Technical Committee Annual Report for 1990. TCCHINOOK (90)-2, Vancouver, British Columbia.**
- PSC (Pacific Salmon Commission). 1988. Chinook Technical Committee Annual Report for 1987, Appendix 2, Supplement B. TCCHINOOK (88)-2, Vancouver, British Columbia.**

- PSMFC (Pacific States Marine Fisheries Commission) Subcommittee on Mass Marking. 1991. Mass marking anadromous salmonids: techniques, options, and compatibility with the coded wire tag system. The Regional Mark Committee, Pacific States Marine Fisheries Commission. Seattle, Washington.
- Petrosky, C.E., and L.B. Everson. 1988. Idaho habitat evaluation for off-site mitigation record. Annual Report 1987. Submitted to Bonneville Power Administration. Contract DE-A179-84BP13381, Project 83-7. Portland, Oregon.
- Petrosky, C.E. 1990. Estimating spring chinook parr and smolt abundance in wild and natural production areas. pp. 57-61. IN: D. Park, Convenor, Status and future of spring chinook salmon in the Columbia River basin – conservation and enhancement. Spring chinook Salmon Workshop, 8-9 November, 1989, Pasco Washington, sponsored by the National Marine Fisheries Service. 130 pp.
- Petrosky, C.E. 1991. Influence of smolt migration flows on recruitment and return rates of Idaho spring chinook. Idaho Department of Fish and Game. Boise, Idaho.
- Petrosky, C.E. and H.A. Schaller. 1992. A comparison of production for Snake River and Lower Columbia River spring and summer chinook stocks. Prepared for Proceedings of Salmon Management in the 21st Century: Recovering Stocks in Decline. 1992 Northwest Pacific Chinook and Coho Workshop, Boise, Idaho.
- Pirtle, R.B. 1957. Field studies to estimate the size and timing of runs of anadromous species of fish in the Columbia and Snake Rivers and their tributaries above the confluence of the Snake River. Idaho Fish and Game. Boise, Idaho.
- Platts, W.S. and W.F. Megahan. 1975. Time trends in riverbed sediment composition in salmon and steelhead spawning areas: South Fork Salmon River, Idaho. Transactions of the 40th North American Wildlife and Natural Resources Conference. Boise, Idaho.
- Poe, T.P., H.C. Hansel, S. Vigg, D.E. Palmer, and L.A. Prendergast. 1991. Feeding of predaceous fishes on out-migrating juvenile salmonids in John Day Reservoir, Columbia River. Trans. Am. Fish. Soc. 120:405-419.

- Pratt, K.L., and D. W. Chapman. 1989. Progress toward the run doubling goal of the Northwest Power Planning Council. Contract report submitted to Pacific Northwest Utilities Conference Committee, Portland, Oregon, 77p.
- Raymond, H.L. 1969. A summary of the 1968 outmigration of juvenile salmon and steelhead trout from the Snake River. U.S. Bureau Commercial Fisheries, Progress Report, Seattle, Washington.
- Raymond, H.L. 1988. Effects of hydroelectric development and fisheries enhancement on spring and summer chinook salmon and steelhead in the Columbia River Basin. North Am. J. Fish. Mangmnt. 8:1-23.
- Ramsey, D. 1969. Northeastern Oregon spring chinook spawning ground surveys for 1969. Fish Commission of Oregon. Clackamas, Oregon.
- Ramsey, D. 1970. Northeastern Oregon spring chinook spawning ground surveys for 1970. Fish Commission of Oregon. Clackamas, Oregon.
- Reisenbichler, R.R. 1990. Sustainable harvest rates for spring chinook salmon in the upper Columbia River Basin. pp. 5-13. IN: D. Park, Convenor, Status and future of spring chinook salmon in the Columbia River basin – conservation and enhancement. Spring chinook Salmon Workshop, 8-9 November, 1989, Pasco Washington, sponsored by the National Marine Fisheries Service. Seattle, Washington. 130 pp.
- Rich, B.A., R.J. Scully, and C.E. Petrosky. 1992. Idaho habitat/Natural Production Monitoring Part I. Submitted to Bonneville Power Administration. General Monitoring Subproject. Portland, Oregon.
- Ricker, W.E. 1972. Hereditary and environmental factors affecting certain salmonid populations, p. 19-160. In: R.D.Simon and P. A. Larkin (eds.). The stock concept of Pacific salmon. H.R. MacMillan lectures in fisheries. Univ. British Columbia, Vancouver, BC.
- Ricker, W.M. 1975. Computation and interpretation of biological statistics of fish populations. Fisheries Research Board of Canada, Bulletin 191, Ottawa.



- Rieman, B.E., R.C. Beamesderfer, S. Vigg, and T.P. Poe. 1991. Estimated loss of juvenile salmonids to predation by northern squawfish, walleyes, and smallmouth bass in John Day Reservoir, Columbia River. *Transactions of the American Fisheries Society* 120:448-458.
- Rondorf, D.W. and W.H. Miller, 1992. Identification of the spawning, rearing, and migratory requirements of fall chinook salmon in the Columbia River Basin. Annual Progress Report prepared for the Lower Snake River Compensation Plan Office, Boise, Idaho, and the Bonneville Power Administration, Division of Fish and Wildlife, Portland, Oregon.
- Olson, F.W. 1984. Vertical distribution of juvenile salmonids entering the turbine intakes at Wanapum Dam. CH2M Hill report to Public Utility District of Grant County. Portland, Oregon.
- Schaller, H. and T. Cooney. 1992. Snake River fall chinook life-cycle simulation model for recovery and rebuilding plan evaluation. In: State and Tribal Fishery Agencies, Chinook Analytical Framework Snake River Fall Chinook Empirical Life Cycle Model Draft Documentation, submitted to the National Marine Fisheries Service, ESA Administrative Record. Portland, Oregon.
- Schiewe, M.H. 1992. Draft 1993 jeopardy criteria. National Marine Fisheries Service, memorandum to Gary Smith, December 23, 1992. Seattle, Washington.
- Swan, G.A., B.H. Monk, J.G. Williams, and B.P. Sandford. 1990. Fish guidance efficiency of submersible traveling screens at Lower Granite Dam - 1989. Annual Report of Research for U.S. Army Corps of Engineers, Contract DACW68-84-H-0034. Portland, Oregon.
- Thompson, G.G. 1991. Determining minimum viable populations under the endangered species act. NOAA Technical Memorandum NMFS F/NWC-198. U. S. Department of Commerce, National Oceanic and Atmospheric Administration. Seattle, Washington.
- U. S. Fish and Wildlife Service. 1990. Colorado Squawfish Recovery Plan. U. S. Fish and Wildlife Service. Denver, Colorado.

- U. S. Fish and Wildlife Service. 1991. Cui-ui (Chasmistes cuius) Recovery Plan (2nd revision). Portland, Oregon.
- United States Fish and Wildlife Service (USFWS). 1991. Snake River hatchery review 1990 workshop summary. Lower Snake River Compensation Plan office. Boise, Idaho.
- Wainwright, T. 1992. Changes in survival needed to meet goals. National Marine Fisheries Service, memorandum to Michael H. Schiewe, December 23, 1992. Seattle, Washington.
- Walla Walla District Corps of Engineers. 1992. Interim Columbia and Snake River flow improvement measures for salmon. Draft Supplemental Environmental Impact Statement. Department of the Army, Walla Walla District, Corps of Engineers. Walla Walla, Washington.
- Wapies, R.S. 1990. Stock structure and gene conservation in Columbia River spring chinook salmon, p. 14-21. In D.L. Park (convener), Status and future of spring chinook salmon in the Columbia River Basin -Conservation and enhancement. U.S. Dep. Commer., NOAA Tech. Memo. NMFS F/NWC-187, National Marine Fisheries Service. Seattle, Washington.
- Waples, R.S., O.W. Johnson, and R.P. Jones Jr. 1991. Status review for Snake River fall chinook salmon. NOAA Technical Memorandum NMFS F/NWC-201. U. S. Department of Commerce, National Oceanic and Atmospheric Administration. Seattle, Washington.
- Washington Department of Fisheries and Oregon Department of Fish and Wildlife. 1992. Status report: Columbia River fish runs and fisheries, 1938-91. Portland, Oregon.
- Warren, C. 1988. 1986 sockeye salmon brood year report. Idaho Department of Fish and Game Special Report Boise, Idaho.
- Weiss, E.F., and R.L. Demory 1963. A survey of Northeastern Oregon Chinook salmon spawning grounds for 1963. Fish Commission of Oregon. Clackamas, Oregon.

- Weiss, E.F., and R.B. Hermann. 1962. A survey of Northeastern Oregon Chinook salmon spawning grounds for 1962. Fish Commission of Oregon. Clackamas, Oregon.
- Young, F. 1969. Northeastern Oregon spring chinook spawning ground surveys for 1968. Fish Commission of Oregon. Clackamas, Oregon.

# APPENDIX 1: ESTIMATING THE PROBABILITY OF EXTINCTION OF POPULATION GROWING OR DECLINING EXPONENTIALLY.

Dennis developed a utilitarian form of expressing the probability of extinction for a population experiencing population growth or decline: The probability of extinction by time C for a specified population size ( $Y_{base}$ ), infinitesimal mean ( $\mu$ ), infinitesimal variance ( $\sigma^2$ ), and a specified extinction number ( $Y_{ext}$ ) is

$$P(T \leq C) = F(z+) + \exp(u)F(z-) \text{ for } \mu \leq 0$$

or

$$P(T \leq C) = \exp(-u)F(z+) + F(z-) \text{ for } \mu > 0$$

wherein

$$u = \frac{2 \ln(Y_{base}/Y_{ext}) + |\mu|}{\sigma^2}$$

$$z+ = \frac{-\ln(Y_{base}/Y_{ext}) + |\mu| \cdot C}{[\sigma^2 C]^{\frac{1}{2}}}$$

$$z- = \frac{-\ln(Y_{base}/Y_{ext}) - |\mu| \cdot C}{[\sigma^2 C]^{\frac{1}{2}}}$$

F(z) being the cumulative normal distribution function evaluated at z.

**NOTE:** There was a typographical error in Dennis et al's 1991 paper for F(z-) (Dennis, 1993, personal communication). The equation given above for P(T≤C) differs slightly in form from Dennis et al's presentations, but the corrections have been incorporated into the above expressions.

The Dennis estimate for the infinitesimal mean and the infinitesimal variance respectively are:

$$\hat{\mu} = \frac{\sum \ln[y(t)/y(t-\Delta t)]}{n\Delta t}$$

$$\hat{\sigma}^2 = \frac{\sum \{\ln[y(t)/y(t-\Delta t)] - \Delta t \hat{\mu}\}^2 / \Delta t}{(n-1)}$$

$y(t)$  and  $y(t-\Delta t)$  being the observed abundance at time  $t$  and  $t-\Delta t$ .

There are two sources of variation that would contribute to the estimated extinction time:

- inherent year-to-year (or generation-to-generation) variation in population growth or decay, and
- variation in the parameter estimate itself.

It is possible to accommodate both sources of variation in estimating the expected extinction probability. If the data variables used to estimate  $\mu$  are independent, then  $S^2(\mu)$ , the variance of  $\mu$ , can be estimated. Recall that the estimate of  $\mu$  was

$$\hat{\mu} = \frac{\sum \ln[y(t)/y(t-\Delta t)]}{n\Delta t}$$

Assuming that  $\ln[y(t)/y(t-\Delta t)]$  is an independent variable over  $t$ , then the estimate of  $S^2(\mu)$  is

$$s^2(\mu) = s^2/n$$

wherein

$$s^2 = \frac{\sum \{\ln[y(t)/y(t-\Delta t)] - \Delta t \hat{\mu}\}^2 / (\Delta t)^2}{(n-1)}$$

**NOTE:**  $s^2$  is not the same as the infinitesimal variance estimate,  $\delta^2$ , given by Dennis, the  $\Delta t$  in the denominator being squared for  $s^2$  but not for  $\delta^2$ ; therefore

$$s^2(\hat{\mu}) = \delta^2 / (\Delta t * n)$$

Letting  $P(T \leq C; \hat{\mu})$  be the probability of extinction for a given estimated mean,  $P(T \leq C)$  could be assessed by integrating  $P(T \leq C; \hat{\mu})$  over the probability density function of  $\hat{\mu}$ . The probability density function can be approximated by substituting the estimates of  $\mu$  and  $\sigma^2(\hat{\mu})$  into the normal probability density function.

## APPENDIX 2: SELECTING THE PROPORTION OF THE POPULATION TO TAKE FOR HATCHERY BROODSTOCK

The proportion of the population to be taken for hatchery broodstock should be chosen so as to maintain the same gene frequency in the hatchery that exists in the population.

The basis for choosing the sampled proportion (s) would be to guarantee that there is an  $1-\alpha$  certainty that the sampled allelic frequency proportionally differs by no more than r from the true population allelic frequency. For  $p \leq 0.5$ ,

$$P\{|\hat{p} - p|/p < r\} = P\{|\hat{p} - p| < rp\} = 1-\alpha \quad [1]$$

using the normal approximation to the binomial,

$$rp = z(\alpha) * SE(\hat{p}). \quad [2]$$

For a finite population, the standard error is approximately

$$SE(\hat{p}) \approx \{ (1-s)[p(1-p)/(2sN)] \}^{1/2} \quad [3]$$

wherein s is the proportion of individuals/alleles sampled; i.e.

$$s = n/N = (2n)/(2N)$$

or

$$n = sN \quad [4]$$

n being sample size and N (Y in the text) being population size of fish.

NOTE: As  $s$  approaches 0 (a small proportion is sampled),

$$SE(p) \approx \{p(1-p)/(2n)\}^{1/2},$$

the usual SE of the allelic frequency of a sample.

Substituting SE of [3] into [2] and solving for  $s$  gives:

$$s = 1/\{f(N, r, p, z(\alpha)) + 1\} \quad [5a]$$

wherein

$$f(N, r, p, z(\alpha)) = 2N[r/z(\alpha)]^2[p/(1-p)] \quad [5b]$$



APPENDIX 3: LOWER GRANITE DAM COUNTS AND RECONSTRUCTED WILD COUNTS (EXCLUDING JACKS) 1975 - 1992.

Spring/Summer Chinook Adult Counts (excluding Jacks)									Fall Chinook Adult Counts (3)		Sockeye Count (1)
	Dam Count (1)			Wild Proportion (2)		Wild Count (2)					
Year	Spring	Summer	Total	Spring	Summer	Spring	Summer	Total	Total	Wild	Wild
1992	(2) 21391	(2) 3008	24399	0.36	0.36	8196	1148	9344	855	533	* 15
1991	6623	3809	10432	0.41	0.74	2708	2014	5520	630	318	8
1220	17315	5093	22408	0.24	0.08	4152	3378	7530	385	78	0
1989	12955	3169	16155	0.30	0.73	3900	2350	6250	708	295	2
1988	29495	6145	35695	0.57	0.37	10915	2263	13178	627	368	23
1987	28835	5891	34735	0.31	0.41	8928	2414	11342	951	253	22
1986	31722	8154	37822	0.25	0.63	7925	3934	11859	784	449	15
1985	25207	5082	30307	0.24	0.63	6048	3198	9244	691	438	35
1984	9500	5400	11900	0.50	0.78	3250	4200	7450	640	324	47
1983	9500	3900	13400	0.05	0.83	6175	3233	9408	540	422	122
1222	12400	4200	16600	0.52	0.64	6448	3529	9977	720	720	211
1981	13100	3300	16400	0.41	1.00	5371	3300	8671	340	340	218
1980	5500	2700	8200	0.40	1.00	2200	2700	4900	450	450	96
1979	8800	2700	9514	0.71	1.00	4828	2714	7542	500	500	25
1978	40700	11800	52300	0.77	1.00	31376	11800	42975	640	640	123
1977	38200	7700	43900	0.04	1.00	23176	7700	30875	M 0	600	458
1976	15900	7000	22900						470	470	531
1975	16100	7600	23700						1000	1000	209

(1) Status Report: Columbia River Fish runs and Fisheries 1938-1991

(2) PSC Chinook Technical Advisory Committee 1991

(3) Laurie Lavoy, Washington Department of Fisheries

(\*) Joanne Kemble, U.S. Army Corps of Engineers

## APPENDIX 4. STATISTICAL TREND MODELS

Model selection should be based on the following criteria:

- a) Are the model parameters easily understood?
- b) Is the model applicable?
- c) Is it easy to fit the data to the model?
- d) Can the variability in the parameter estimates be characterized?
- e) Can the parameters be easily tested statistically?
- f) What are the relative biases and precisions of the parameter estimates?
- g) Is it possible to adjust for factors (covariates) that are  
not functions of the recovery program, and  
likely to affect abundance or trend?
- h) is it possible to characterize 'random' variability?

### A4.1 EXPONENTIAL TREND MODELS

Exponential trend models are of the form

$$y(t) = A \cdot \exp(B \cdot t) \cdot e$$

in which the spawner abundance measure ( $y$ ) at time  $t$ , is a function of time, the abundance ( $A$ ) at time  $t = 0$ , the instantaneous rate ( $B$ ) of growth or decline, and  $e$ , a multiplicative error "exp" is the exponential constant.

- a) The parameters can be easily understood. Instantaneous exponential growth ( $B > 0$ ) or decline ( $B < 0$ ) is a measure on a per-unit time basis. The actual change

is in the form of a proportion decline or growth

$$P = 1 - \exp(B) \text{ for } B < 0 \text{ (decline)}$$

or

$$P = \exp(B) - 1 \text{ for } B > 0 \text{ (growth).}$$

- b) The model may be applicable to time periods of decline (the historical record) or accelerating growth (the early part of the recovery period). At a point where the spawner density of this population is high enough that the growth rate begins slowing down, this model will no longer be appropriate.

#### A4-1.1 Simple regression of $\ln(y)$ on $t$

The natural log transformation of the exponential model gives

$$\ln(y) = \ln(A) + B \cdot t + e$$

wherein  $e = \ln(e)$

where  $e$  is normally distributed and the  $e$ 's are independent.

- c) The model can be readily fit using least squares regression programs available in any decent statistical package.
- d) Any decent statistical package will print out estimates of the standard errors (SE) of the estimates.
- e) Statistical tests of the form

$$t = \frac{\text{estimate} - \text{hypothesize value}}{\text{SE}(\text{estimate})},$$

can be computed from the computer output. Most statistical packages will print out a t-value associated with a hypothesized value of 0 and a probability that

corresponds to a critical region defined by the computed  $t$ .

- g) Estimates of  $\ln(A)$  will be unbiased, but the estimate of  $A$  based on the re-transformation of the estimate of  $\ln(A)$  will be biased. This bias can distort the plot of the retransformed  $t$ .

The estimate of  $B$  can also be biased if the abundance measure is ever equal to 0, which does happen for redd counts. It is not possible to take the log of 0. What is usually done is either to drop the data point from consideration or to add 1 to all of the data sets and perform the regression on  $\ln(y+1)$ . Either of these techniques will lead to biased estimates of  $B$ . We will discuss the possible biases associated with adding 1 later.

If there are autocorrelations over time, then  $B$  would be better estimated if the autocorrelations could be adjusted for. There are likely to be two sources of autocorrelation:

- One source of autocorrelation would be due to environmental similarities among contiguous years. It may be possible to effectively reduce this source by adjusting for appropriate environmental covariates.
- The other source would be due to return-year to brood-year associations. Returns from adjacent years would have shared part of their brood-year. Correlation between adjacent years (referred to in statistics texts as first-order autocorrelation) can be tested for using the Durbin and Watson test. There could also be correlations associated with returns of a given year and the returns of the given year's brood-stock. These correlations would be more difficult to test for. If the correlations are known, then a weighted least squares is appropriate, the weight being the inverse of the correlation matrix.

If the variances of  $\ln(y)$  for a given year are not equal (homogeneous) over years, then  $B$  may not be poorly estimated. However, this is not likely to be a serious problem. The variance of the untransformed abundance variable would be expected to have a Poisson distribution. If this is the case, then the log transformed variable will tend to have a homogeneous variance over time.

- g) It is possible to adjust parameter estimates for a covariate by including the covariate as an additional explanatory variable (z):

$$x = \ln(y) = \ln(A) + B*t + D*z + e$$

- h) The mean square error (MSE) generated from the analysis can be used to characterize the random variability.

A4.1.2. **Compute the mean of the log of ratios**

This is the same estimate that Dennis uses for assessing extinction probabilities (Appendix A). The estimate is of the form

$$b = \text{mean}(z)/\Delta t$$

wherein

$$z(t) = \ln[(y(t))/(y(t - \Delta t))]$$

mean(z) being the mean of the z(t)'s over t.

- c) The mean is readily estimated.
- d)  $SE(b) = SE[\text{mean}(z)]/\Delta t$ .
- e) The statistical test is of the form

$$t = (b - \text{hypothesize value})/SE(b)$$

- f) The estimate of B can be biased if the abundance measure is ever equal to 0, which does happen for redd counts. Dropping a 0 will usually lead to the loss of at least two data points. A drop of a numerator 0 for return year would also lead to dropping a denominator 0 in the next generation for which the 0 applies to a brood year. If, rather than dropping the 0, 1 is added, it must be added to all

counts, both denominator and numerator counts. We will discuss the bias resulting from adding 1 later.

Regarding the two likely sources of autocorrelation, environmental and brood-year-based correlations, the same kind of weighted least squares approaches discussed under the simple regression estimate can be applied to the mean. The mean can be estimated using regression model of the form

$$z = C*v + e$$

where no intercept is fit (an option available in any decent regression package). The  $v$  values are all set equal to 1. When  $z$  is regressed on  $v$ , the estimate of  $C$  will be the mean of the  $z$ 's. When the regression is weighted by the inverse of the correlation matrix,  $\text{mean}(z)$  will not be altered but the  $\text{SE}(z)$  will be; remember  $b = \text{mean}(z)/\Delta t$ ,  $\text{SE}(b) = \text{SE}[\text{mean}(z)]/\Delta t$ .

One source of autocorrelation mentioned for simple linear regression, that between a given year's returns and the returns of the given year's brood-stock, will not be important for the log ratio estimate.

The relative precision associated with the log-ratio estimate is expected to be less than that associated with simple linear regression of  $\ln[y]$  on  $t$ . This is because

$$z(t) = \ln[(y(t))/(y(t-\Delta t))] = \ln[y(t)] - \ln[y(t-\Delta t)]$$

and since  $z$  involves the difference of two  $\ln[y]$ 's, its variance is expected to be twice that of  $\ln[y]$  in the absence of autocorrelation.

- g) It is possible to adjust parameter estimates for covariates by using the regression form of the model given under f) and including the covariate as an explanatory variable ( $z$ ):

$$z = C*v + D*z + e$$

Again, the adjustment is expected to reduce the SE but not affect the mean.

- h) The mean square error (MSE) generated from the analysis can be used to characterize the variability. Since the estimate of B is

$$b = \text{mean}(z)/\Delta t$$

The appropriate MSE associated with b is

$$\text{MSE}(b) = \text{MSE}(z)/(\Delta t)^2$$

**NOTE:** The above estimate of B is the same as the infinitesimal mean of Dennis, however, MSE(z) is not the same as the infinitesimal variance given by Dennis (see Appendix A).

#### **A4.1.3 Poisson regression of y on t**

This is a generalized linear regression technique that does not require transforming the data. It is not a least squares technique. It assumes the multiplicative model and assumes that the abundance has a Poisson distribution, which is a reasonable assumption for survival probabilities. It is expected to give unbiased or nearly unbiased estimates of both A and B.

- c) The model can be readily fit using appropriate statistical packages. However, many statistical packages do not have a Poisson regression program.
- d) Most statistical packages that have Poisson regression programs will print out asymptotic estimates of the standard errors (SE) of the estimates. These SEs are usually too small.
- e) The most common statistical measure used for hypothesis testing is the t-ratio,

$$t = \frac{\text{estimate} - \text{hypothesis value}}{\text{SE}(\text{estimate})}.$$

the t-ratio being assessed as to whether it fell outside of some probability-based

critical region when the hypothesized value was the true value.

The asymptotic SE printed by computer packages is usually too small an estimate for this test and the probability printed out by these packages will be too small. (The t-tests come out significant far more often than they should.)

The appropriate statistical test would be the log likelihood test. The only reasonably priced statistical package that we are aware of and that permits appropriate statistical tests is GLIM (General Linear Interactive Modeling distributed by NAG, Downers Grove, Illinois at \$588 for DOS 3.0). Several other cheaper packages perform Poisson regressions but do not give the appropriate log likelihood test.

- f) The Poisson regression estimates may be the least biased overall because they characterize the actual distribution. The other estimates utilizing the log-transformation use the log-normal distribution to approximate the actual distribution on the transformed scale:

Estimates of both A and B are expected to be unbiased or reasonably unbiased.

The model is directly based on what is likely to be the best assumed distribution of abundance

- The model can be used when the abundance measure is 0

Since the SE printed out by Poisson regression programs is approximate, it cannot be compared to those associated with the simple linear regression of  $\ln(y)$  on  $t$  or with the mean of the log-ratio.

- g) The program can be used to adjust parameter estimates for covariates such as drought indices.
- h) The program can be used to characterize the variability either by using residuals or by using a pseudo-MSE (MSE - mean square error) based on a measure referred to as deviance that is analogous to the error sums of squares from least squares regression programs. These measures apply directly to the abundance



measure, not to the log transformation of the abundance.

#### A4.1.4 **Comparison** of the three exponential trend estimators<sup>s</sup> in presence of **zero counts**

There are many cases of zero redd counts from several of the redd index areas. We performed analyses using Snake River spring-summer chinook redd count data from the eightdam era, 1978-1992, to determine which of the three estimation methods to follow for trend assessment of individual index areas. One was added to all counts before log transformations were made; however, the data used for the Poisson regression were not altered. The estimates of the Poisson are assumed to be unbiased, or nearly so; if the other methods did not differ greatly from those of the Poisson, we would have opted for one of the other methods because they are more familiar to biologists. Such was not the case.

Regressions were fit for only those index areas having counts for every year within the period. The estimates are given in Table A4-1.

As expected, the SE of the log-ratio was greater than that of the simple linear regression of logs, indicating the greater precision of the simple linear regression.

Regarding potential bias resulting from adding one to data points, both the log ratio and the simple linear regression based on logs estimated less decline (or more growth) than did the Poisson regression in 19 out of 24 index areas. This and the fact that the average bias was quite large led us to believe that the Poisson regression should be applied in the presence of zero counts.

The probabilities associated with the Poisson test were based on a pseudo-F-test

$$F = \frac{\text{Deviance(fitting A)} - \text{Deviance (fitting A and B)}}{\text{Deviance(fitting A and B)/(n-2)}}$$

n being the number of years.

Table A4-1 Different estimates of exponential trend when there are zero counts in the data

Subbasin	Index Area	ESTIMATES									BIAS MEASURES*		
		Log Ratio (LR)			Simple Linear Regression			Poisson Regression		Deviation of means	(LR-PR)	(SLR-PR)	
		Mean	SE	P	Mean	SE	P	Mean	P**				
		$\ln(c_i + 1)/(c_i + 1) / 5$			$(SLR \text{ of } \ln(c_i + 1) \text{ on } t)$			$(PR \text{ of } c_i \text{ on } t)$					
Grande Ronde	Willows Cr.	-0.0312	0.0885	0.78	-0.0418	0.0859	0.54	-0.0755	0.39	0.0443	0.0338		
	Bear Cr.	-0.1011	0.0728	0.30	-0.1089	0.0408	0.02	-0.1178	0.02	0.0188	0.0107		
	Wenaha Cr.	0.0413	0.0543	0.47	0.0470	0.0474	0.34	0.0318	0.47	0.0084	0.0151		
	Hurricane Cr.	-0.0385	0.1182	0.74	-0.0159	0.0700	0.82	-0.0280	0.71	-0.0105	0.0131		
	Lootline Cr.	-0.0271	0.0788	0.74	-0.0475	0.0482	0.34	-0.0501	0.29	0.0230	0.0026		
Imnaha	Big Sheep Cr.	-0.0403	0.1020	0.70	-0.0567	0.0883	0.38	-0.0582	0.40	0.0179	-0.0015		
	Lick Cr.	-0.2124	0.0715	0.02	-0.2110	0.0438	<0.01	-0.5122	<0.01	0.2998	0.3013		
Salmon	Imnaha R.	-0.0489	0.0880	0.41	-0.0525	0.0381	0.17	-0.0835	0.05	0.0338	0.0310		
	Altura/Lake Cr.	-0.1870	0.0848	0.05	-0.2089	0.0801	<0.01	-0.5073	<0.01	0.3103	0.2884		
	Lameli Cr.	-0.0838	0.0888	0.31	-0.0871	0.0530	0.09	-0.1450	0.02	0.0512	0.0478		
	Upper Salmon R.	-0.0779	0.0883	0.29	-0.0984	0.0502	0.07	-0.2372	<0.01	0.1598	0.1378		
	Upper Valley	-0.0880	0.1209	0.43	-0.1218	0.0723	0.12	-0.2438	0.01	0.1455	0.1220		
	Bear Valley Cr.	0.0028	0.0750	0.87	-0.0135	0.0482	0.78	-0.0102	0.84	0.0128	-0.0033		
	Elk Cr.	0.0858	0.0855	0.34	0.0533	0.0565	0.39	0.0185	0.78	0.0881	0.0338		
	Mason Cr.	0.0301	0.0875	0.74	0.0181	0.0502	0.72	-0.0208	0.87	0.0510	0.0389		
	Sulphur Cr.	0.0310	0.1035	0.77	0.0182	0.0748	0.83	-0.0158	0.82	0.0488	0.0319		
	Lower Valley Cr.	-0.0481	0.0884	0.60	-0.0781	0.0882	0.27	-0.1771	0.03	0.1280	0.0991		
	Johnson Cr.	0.0443	0.0385	0.29	0.0347	0.0318	0.39	0.0242	0.45	0.0301	0.0105		
	South Fork	0.1184	0.0433	0.02	0.1122	0.0279	<0.01	0.1100	<0.01	0.0084	0.0022		
	Clearwater	Selway R.	-0.1114	0.0448	0.03	-0.1052	0.0325	<0.01	-0.1245	<0.01	0.0130	0.0183	
Crooked R.		-0.0215	0.0732	0.78	-0.0085	0.0397	0.81	-0.0143	0.88	-0.0072	0.0048		
Brushy Fork Cr.		-0.1354	0.0489	0.02	-0.1344	0.0284	<0.01	-0.1182	<0.01	-0.0172	-0.0182		
Newson Cr.		-0.1347	0.0887	0.18	-0.1487	0.0512	0.01	-0.0851	0.11	-0.0398	-0.0538		
Red R.		-0.0228	0.0885	0.75	-0.0218	0.0438	0.63	-0.0031	0.94	-0.0188	-0.0187		
Summary for mean of coefficients over 24 index areas													
	Mean	-0.0454	0.0775		-0.0538	0.0504		-0.1022		0.0568	0.0463		
	SE(Mean)	0.0188			0.0188			0.0308		0.0188	0.0181		
	t-ratio	-2.7112			-3.2418			-3.3187		3.0472	2.8778		
Summary for coefficient estimate based on pooled count over 24 index areas													
		-0.0020	0.0530	0.87	-0.0108	0.0381	0.77	-0.0485	0.28	0.0448	0.0380		
Summary for mean of coefficients over 23 index areas (excluding S.Fork)													
	Mean	-0.0828	0.0780		-0.0811	0.0514		-0.1114		0.0589	0.0504		
	SE(Mean)	0.0158			0.0158			0.0307		0.0183	0.0187		
	t-ratio	-3.3030			-3.8088			-3.8287		3.0513	2.6888		
Summary for coefficient estimate based on pooled count over 24 index areas (excluding S.Fork)													
		-0.0388	0.0561	0.51	-0.0408	0.0377	0.22	-0.0827	0.08	0.0540	0.0429		

\* Poisson regression estimator assumed to be unbiased, log estimator of count + 1 assumed to be biased

\*\* Poisson probability based on pseudo-F ratio of difference in deviances to deviance of full model divided by degrees of freedom

Biases exist in the absence of adding one when simple linear regression predictions based on the log are retransformed to abundance counts. This is due to the bias in the estimate of A. This bias is evidenced in the plots in Figure A4-1 of simulated data and fits based on the Poisson and based on the re-transformed simple regression fit based on logs (logarithm). The retransformed counts are much higher than the actual counts toward the beginning of the exponential decline and much closer to the actual toward the end; whereas the Poisson gives a better fit overall, although it tends to overestimate toward the end of the decline for this example.

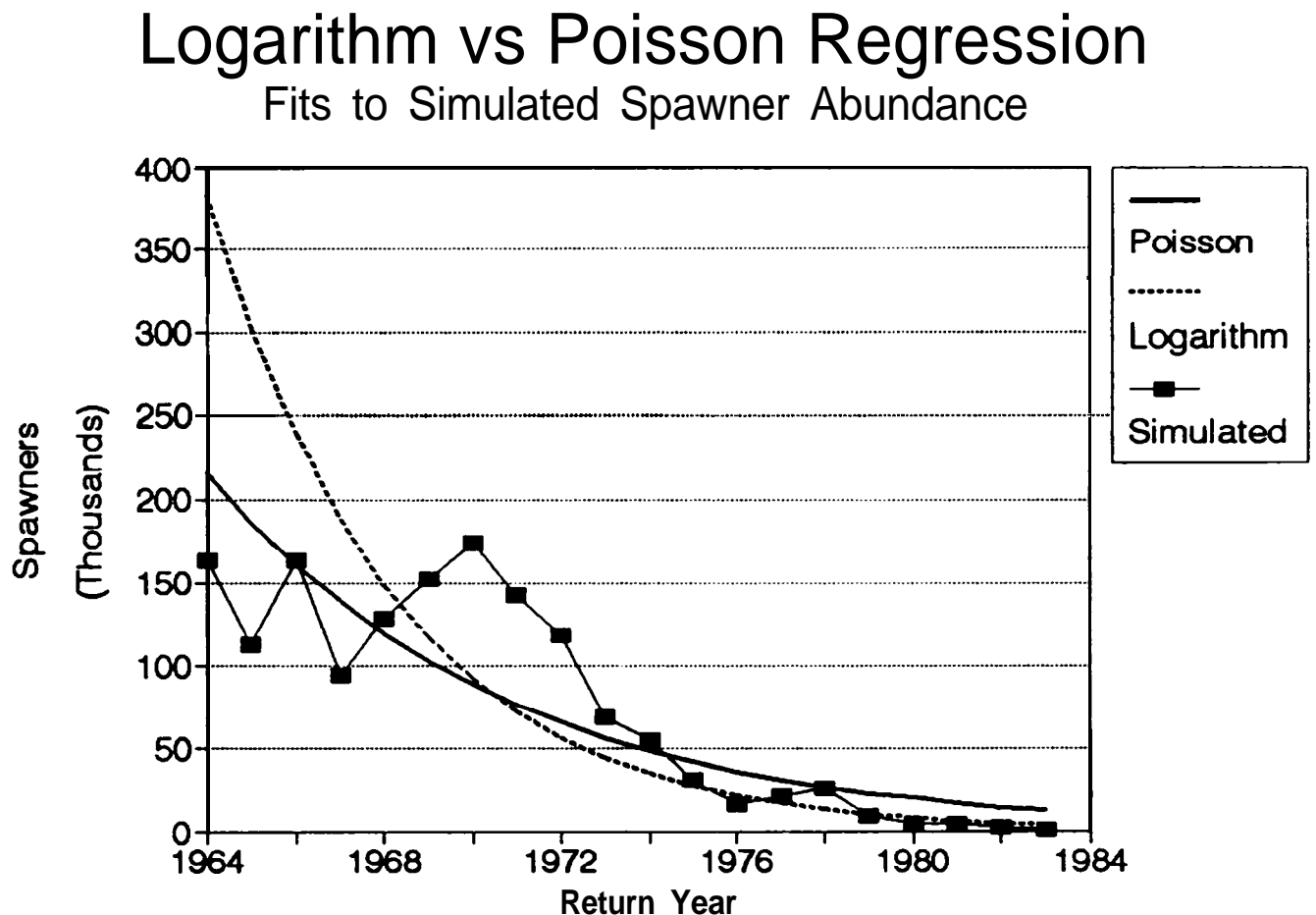


Figure A4-1 Retrtransformed fits of log-linear regression versus Poisson regression fits to simulated data.

## A4.2. THE RICKER FUNCTION

The Ricker function takes the form

$$y(t) = y(t-\Delta t) \cdot \alpha \cdot \exp[-\beta \cdot y(t-\Delta t)] \cdot e$$

$$= y(t-\Delta t) \cdot \exp[a - \beta \cdot y(t-\Delta t)] \cdot e$$

- a) The parameters can be understood

Point at which rate of growth decreases =  $1/\beta$   
(maximum production)

Carrying capacity =  $\ln(\alpha)/\beta$   
(replacement value)

- b) The model is applicable to a stable environment.
- c) It is easy to fit the model using simple linear regression based on the following transformed model:

$$\ln\{y(t)/y(t-\Delta t)\} = a - \beta \cdot y(t-\Delta t) + e$$

The same assumptions that generally apply to least squares estimates (e.g., independence and homogeneity of variance) apply here as well.

- d) The variability in the parameter estimates as measured by SE are part of standard regression output; however  $SE(\beta)$  will not be appropriate for testing hypotheses concerning  $\beta$  using standard t-tests.
- e) Statistical tests for  $\beta$  are not standard because the response variable,  $\ln[y(t)/y(t-\Delta t)]$ , includes the explanatory variable,  $y(t-\Delta t)$ ; i.e.  $y(t-\Delta t)$  occurs on both sides of the regression equation.

For the return-spawner-recruit to brood-year-spawner-parent ratio, the test

recommended by Dennis (1993) can be made in the following fashion, first fit the model

$$\text{Model 1: } z(t) = a$$

which is used in the log-ratio mean estimate of the exponential growth rate. Test whether

$$H_0: a = 0$$

$$H_1: a \neq 0$$

This test is the usual t-test for the mean

$$t = \frac{a}{SE(a)}$$

Using this model as a base, then test

$$\text{Model 2: } z(t) = a - \beta * y(t - \Delta t)$$

At this point non-traditional statistical tests have to be made. Dennis developed a parametric boot-strap technique for testing

$$H_0: \beta = 0$$

$$H_1: \beta > 0$$

for the return-spawner (recruit) to brood-year spawner (parent) ratio. Compute

$$t = \frac{\beta}{SE(\beta)}$$

but do not compare it to values from a t-table. Dennis recommends the following one sided test for

$$H_0: \beta = 0$$

$$H_1: \beta > 0$$

(a negative  $\beta$  has no biological meaning). If the estimate of  $\beta$  is negative, do not accept  $H_1$ . If it is positive, then use the following bootstrap technique for the test

Generate thousands of data-set runs using the estimate from the first model,  $z(t) = a$ , as a parameter with a normally distributed error on the log scale, and use the estimated returns as the brood-year spawners to generate the next generation.

Fit the model for each run

For each fit, compute the t-ratio

Rank the generated t-ratios and truncate the upper one-tailed proportion to establish the critical value.

Compare the original t-ratio to the critical value and accept  $H_1$  if the t-ratio exceeds this value.

- f) During the initial generations when the growth rate is increasing in an exponential fashion, the statistical test associated with  $\beta$  will not be very powerful, and negative estimates of  $\beta$  are possible (even though biologically meaningless). This is equivalent to saying the precision associated with the estimate of  $\beta$  is low during the phase of exponential growth.
- g) It is possible to adjust for covariates.
- h) It is possible to characterize "random" variability.

It may be possible to use maximum likelihood non-linear fitting programs to estimate the parameters from the original model

$$y(t) = y(t-\Delta t) \cdot \alpha \cdot \exp[-\beta \cdot y(t-\Delta t)] \cdot e$$

without dividing by  $y(t-\Delta t)$  and taking the log transformation.

# APPENDIX 5: ANNUAL SPRING/SUMMER CHINOOK REDD COUNTS WITHIN GRANDE RONDE AND IMNAHA SUBBASINS.

Year	Grande Ronde Subbasin					Imnaha Subbasin		
	Wallowa River	Bear Creek	Wenaha River	Hurricane Creek	Lostine River	Lick Creek	Big Sheep Creek	Imnaha River
1992	0	0	58	1	14	0	0	75
1991	1	2	28	4	11	1	1	51
1990	0	2	31	0	16	0	0	43
1989	0	2	9	2	20	0	1	40
1988	7	5	98	0	107	0	14	135
1987	15	10	62	17	49	0	3	112
1986	7	10	68	5	48	2	15	127
1985	3	6	36	20	66	3	6	145
1984	0	11	12	0	56	2	7	119
1983	5	6	23	7	39	0	11	95
1982	1	12	27	8	58	2	9	129
1981	0	2	20	1	8	4	2	99
1980	1	4	24	0	18	4	0	40
1979	0	4	5	0	21	32	0	52
1978	18	25	77	11	120	32	14	415
1977	2	12	60	0	25	5	5	143
1976	15	17	20	0	77	17	24	127
1975	1	33	30	2	33	11	14	149
1974	7	21	49	11	117	12	8	277
1973	11	16	205	10	138	16	31	366
1972	5	55	71	18	127	27	28	366
1971	12	30	164	23	82	13	57	366
1970	14	25	279	17	80	50	55	176
1969	17	23	254	9	108	4	30	302
1968	11	40	128	20	126	34	36	241
1967	15	11	185	3	177	30	30	215
1966	14	12	278	1	187	47	61	223
1985	32	15	79	17	102	25	26	189
1964	35	24	167	28	335	14	40	260
1963	41	0	186	33	97	0	0	133
1962	0	0	0	0	53	0	0	248
1981	0	0	0	0	44	0	0	221
1986	0	0	0	0	95	0	0	323
1959	0	0	0	0	0	0	0	0
1958	0	0	0	0	0	0	0	0
1957	0	0	0	0	0	0	0	0
1956	0	0	0	0	0	0	0	0

Sources: ODFW Wallowa District Annual Report 1992  
Rhine Messmer (ODFW) 1993



# APPENDIX 6: ANNUAL SPRING/SUMMER CHINOOK REDD COUNTS WITHIN SALMON AND CLEARWATER SUBBASINS.

Year	Salmon River Drainage Wild Spring Chinook Redds					Salmon River Drainage Wild Summer Chinook Redds				
	Bear Valley	Elk Creek	Marsh Creek	Sulphur Creek	Upper Big	Loon Creek	LakeCr. Secesh R	Lower salmon	Lower Valley	Lower East Fk.
1992	41	57	65	5	22	10	125	26	6	26
1991	47	54	40	26	13	5	112	68	3	23
1990	62	42	57	22	20	***	55	52	9	19
1989	15	35	44	2	30	16	78	77	26	51
1956	283	330	217	41	101	5	155	148	33	85
1967	102	149	150	11	36	23	121	193	59	62
1986	74	55	101	65	67	21	115	104	16	41
1985	134	28	106	10	70	26	105	82	1	9
1964	55	27	60	0	42	4	***	51	15	7
1963	56	38	33	8	27	7	98	111	28	27
1982	39	9	40	3	7	23	65	39	6	19
1951	60	23	63	7	22	30	53	75	17	43
1980	15	8	9	2	***	9	20	11	4	0
1979	69	49	47	15	15	***	20	• □□	15	33
1978	184	208	270	64	95	29	91	□ □ □	219	• □□
1977	129	85	98	5	9	62	27	□ □ □	63	136
1976	76	61	48	14	22	31	17	□ □	43	39
1975	215	169	201	50	77	32	10	□ □	80	38
1974	130	108	210	30	28	47	21	***	45	49
1973	387	375	518	78	95	76	62	224	77	136
1972	221	212	312	71	60	150	87	412	39	161
1971	106	173	281	58	32	79	60	220	147	149
1970	334	302	456	93	68	43	63	150	41	123
1969	356	349	222	138	65	110	104	120	22	138
1968	574	483	466	142	90	135	58	223	63	235
1967	445	420	650	134	67	164	140	365	79	234
1966	534	525	406	142	123	49	140	390	184	216
1965	301	203	404	32	73	166	134	201	57	131
1964	576	425	709	49	51	361	181	415	71	306
1963	460	654	372	140	148	261	163	195	50	265
1962	484	426	341	78	223	157	261	467	115	195
1961	675	581	526	121	377	131	191	356	156	559
1960	386	346	299	39	155	334	510	811	137	403
1959	381	458	88	41	86	123	246	352	70	246
1956	312	359	262	131	129	193	355	460	47	345
1957	661	398	458	381	225	425	328	2533	331	656

\*\*\* Data missing, unclear, or inconsistent

(Source: P. Hassemer, IDFG, Boise, personal communication)

## APPENDIX 6 (CONT.)

	m m				- - W - -				
	Spring Chinook Redds				influenced Summer Chinook Redds				
Year	Alturas Lake Creek	Lemhi River	Upper East Fork	Upper Salmon	Upper Valley	Upper Yankee	Johnson	South Fk.	TOTAL
1992	2	15	6	51	1	10	76	685	761
1991	3	55	21	83	2	0	64	393	457
1990	0	80	***	97	3	3	56	386	442
1959	7	32	***	101	23	7	42	217	259
1988	1	179	***	136	12	1	137	718	855
1987	9	155	***	151	31	0	72	752	624
1986	14	157	***	122	13	15	53	289	342
1985	7	93	***	120	1	5	75	323	396
1984	3	35	***	71	6	***	17	165	162
1983	27	46	121	161	8	0	63	185	248
1962	9	149	28	42	1	0	37	111	148
1951	4	115	76	404	2	4	45	126	171
1980	7	25	6	47	6	0	24	116	140
1979	29	146	57	205	25	18	36	115	151
1978	303	703	841	1707	141	33	113	251	364
1977	85	463	168	698	18	6	81	226	307
1978	16	227	75	378	***	40	68	241	309
1975	60	365	348	509	189	60	69	238	367
1974	42	237	346	338	127	54	107	218	325
1973	153	433	685	411	125	104	271	586	857
1972	143	473	448	748	162	115	220	567	787
1971	50	392	370	619	89	57	163	421	604
1970	66	344	468	432	202	67	130	527	667
1969	41	328	174	313	35	53	273	636	909
1968	110	572	622	637	330	234	127	515	642
1967	74	786	614	943	253	250	285	902	1166
1966	119	738	511	561	219	112	110	980	1090
1965	101	433	136	472	204	77	116	656	772
1964	80	1036	405	706	199	146	310	1124	1434
1963	86	364	646	638	141	128	266	1057	1323
1962	138	1309	334	638	157	60	295	1589	1884
1961	30	1720	618	723	227	192	201	1056	1259
1960	33	1262	122	579	87	43	486	2177	2663
1959	18	468	75	486	23	10	278	1305	1583
1958	96	555	141	469	63	38	225	1206	1431
1957	110	719	61	1101	219	47	319	2732	3051

## APPENDIX 6 (CONT.)

Year	Clearwater River Drainage Natural Chinook Redds					Clearwater River Drainage Hatchery-influenced Chinook Redds					
	Selway River	Bear Creek	Running Creek	Whitecap Creek	Moose Creek	Lochsa River Drainage		Newsome Creek	South Fork Drainage		
						Crooked River	Brushy Fork		Crooked River	Red River	American River
1992	16	9	0	0	2	22	1	0	***	46	1
1991	12	6	0	1	2	9	1	0	• 24	5	1
1990	13	6	1	2	2	16	4	0	10	66	2
1989	5	7	0	3	3	6	9	4	3	45	1
1988	36	10	2	5	7	42	9	20	27	51	12
1987	36	9	4	6	6	26	10	15	17	61	31
1986	30	10	• • •	7	9	30	11	6	9	62	14
1985	15	***	• 24	***	• • •	47	14	7	10	92	23
1984	30	6	***	6	7	26	9	1	22	65	***
1983	26	6	• 24	4	6	7	6	7	12	65	9
1982	36	6	24 24	3	5	34	17	5	4	62	21
1981	47	6	***	4	6	27	25	7	9	47	12
1980	40	7	1	3	4	16	10	7	6	31	7
1979	21	3	0	2	4	6	12	9	4	20	• • •
1976	125	13	6	• 24	17	37	25	22	17	52	• • •
1977	97	16	2	1	23	51	15	26	21	50	• • •
1976	58	14	3	4	15	33	13	5	13	15	***
1975	21	5	• • •	1	4	22	4	6	33	20	• • •
1974	66	10	4	2	15	22	6	***	5	12	***
1973	261	26	21	7	32	60	• • •	• • •	• • •	***	• • •
1972	175	25	11	6	13	32	31	• • •	***	• • •	• • •
1971	55	14	6	***	• • •	1	***	***	• • •	• • •	• • •
1970	65	19	10	4	• • •	34	***	• 24	• • •	• • •	• 24
1969	57	6	21	***	***	112	***	***	***	• 24	• 24
1968	16	7	4	• • •	• 24	15	• • •	***	***	24 24	24 24
1967	22	7	• • •	• • •	• 24	0	***	***	• • •	***	24 24
1966	36	6	• • •	• 24	***	***	• 24	***	***	***	***
1965											
1964											
1963											
1962											
1961											
1960											
1959											
1958											
1957											